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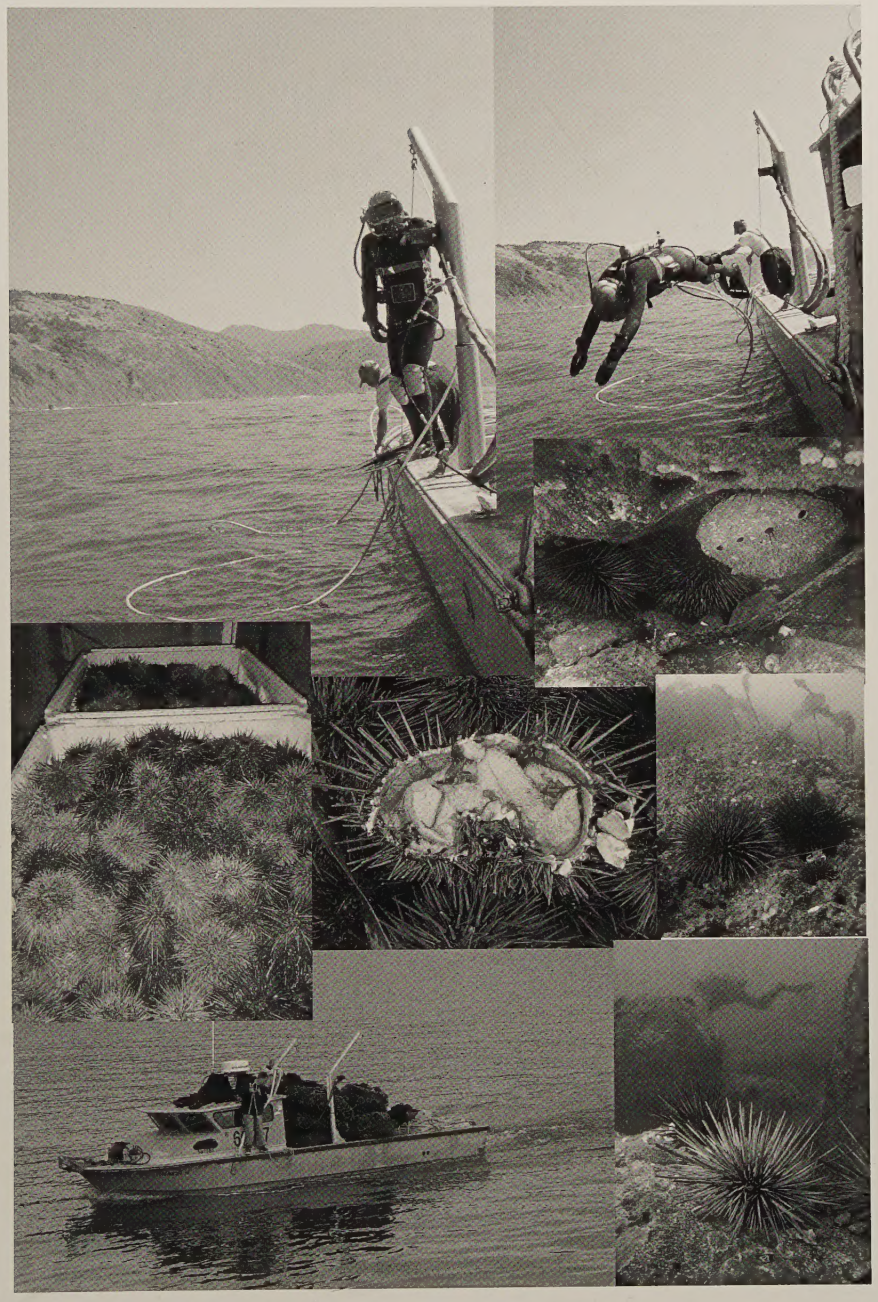

Oregon Sea Urchin Fishery

Marine Fisheries REVIEW

W. L. Hobart, Editor
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On the cover: The Oregon sea urchin fishery. Photos: Scott Groth, ODFW.

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History and Trends in Oregon's Red Sea Urchin, *Mesocentrotus franciscanus,* and Purple Sea Urchin, *Strongylocentrotus purpuratus,* Fisheries, 1986–2022

SCOTT GROTH

Introduction

Biology

The red sea urchin, *Mesocentrotus franciscanus,* is the largest species of sea urchin (up to 211 mm test diameter (TD)) common to the rocky nearshore of the U.S. west coast (Ebert et al., 2018). It ranges from Baja Mexico to Kodiak, Alaska (Ebert et al., 1999) where the species is common in shallow (i.e., 3–35 m depth) rocky reef habitats and most abundant within kelp beds (Rogers-Bennett, 2007). They eat marine vegetation, such as drifting kelp. When well fed their gonads become robust in winter months in preparation for spring spawning (Dean et al., 1984).

Reproduction is episodic, affected by coastal circulation and upwelling conditions (Morgan et al., 2001), and consequently they have evolved life spans greater than 100 years (Ebert and Southon, 2003). Reproduction may also be affected by high water temperatures which shorten their larval period (Cameron and Schroeter, 1980) and may lead to conditions of higher predation and greater stress (Russell, 1987). While their larvae may travel great distances during their 62–131 day larval period (Strathmann, 1978), once settled, they only travel short distances, mostly when food is scarce (Mattison et al., 1977).

The red sea urchin's complex biological characteristics demand skillful fishing since the fishery product is their gonads, the quality of which depends heavily on seasonal, environmental, and demographic conditions. Sea urchin gonads are marketed as "uni," typically combined with rice to make a popular sushi item.

Fishery History

Red sea urchin commercial fishing along the U.S. west coast is relatively recent and has undergone major shifts. The fishery began in the 1970's in southern California, and by the 1980's market demand accelerated and exploration for unexploited stocks extended throughout their range (Kato and Schroeter, 1985). Between the late 1980's and early 1990's Japan's economy expanded which increased demand for uni and stimulated the fishery (Kalvass and Hendrix, 1997). Broadly, the west coast red sea urchin fishery was characterized by a rapid increase in landings, then an equally rapid decrease as stocks were depleted (mid-1990's), followed by a long period (1995–2022) of fishery stability (Fig. 1). State-specific red sea urchin fishing histories along the U.S. west coast were reviewed for northern California by Kalvass and Hendrix (1997), for Washington by Pfister and Bradbury (1996), and for Oregon in this paper. Little recreational catch of sea urchins occurs on the U.S. west coast (personal observ.).

In 1986, the search for unexploited red sea urchin stocks reached Oregon. At that time, the red sea urchin's primary predator, the sea otter, *Enhydra lutris,* had been extirpated by fur traders from Oregon more than a century prior (Jameson et al., 1982). Lacking a predator or fishery pressure, red sea urchin densities were likely at a historic peak. High-density stocks were indeed discovered and infrastructure (e.g., vessels, processing plants,

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ABSTRACT—The red sea urchin, Mesocentrotus franciscanus, *is a large echinoid common to the rocky vegetated nearshore areas throughout the U.S. west coast. Red sea urchin fisheries are among the U.S. west coast's most valuable dive fisheries. These fisheries are broadly characterized as "boom and bust," where a virgin stock is quickly removed without much replacement via recruitment. The unique life history of red sea urchin (long lived, episodic recruitment, etc.) deeply affected the patterns of the stock and consequently the fishery.*

Oregon's red sea urchin fishery also experienced a boom and bust, like many similar fisheries. Fishing began in 1986 and boomed to total catches greater than 4,200 t in 1990. By 1993, the catch dropped below 1,000 t per year where it has remained. Between 1997 and 2017, catch remained low but value was steady, characterized by a low number of participants fully involved in the fishery. During 2018–22, catch has remained steady; however, unit price grew five times the previous era. For years, Oregon's (previously unfished) stock of red sea urchin was removed by the fishery without new recruitment and did not operate in a way consistent with classic definitions of "sustainable." While stock levels were the lowest on record in 2014, ensuing episodic recruitment events were so great that densities of red sea urchin returned to unfished levels. The future of Oregon's red sea urchin fishery likely depends heavily on episodic recruitment events which occur infrequently. Managing the stock and fishery then is complicated and complex and may depend on long-term monitoring combined with a high degree of management flexibility.

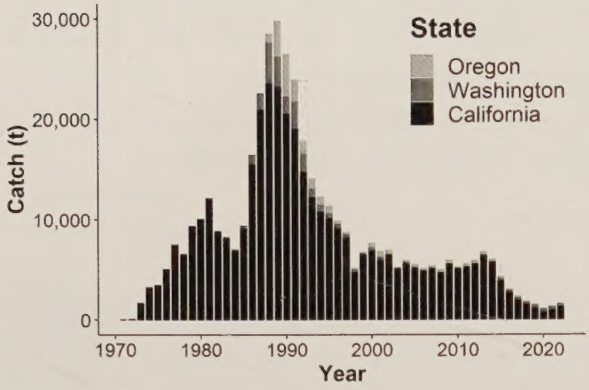


Figure 1.—Red sea urchin landings in Oregon, Washington, and California by year, 1972–2022 (Pacific States Marine Fisheries Commission, 2022).

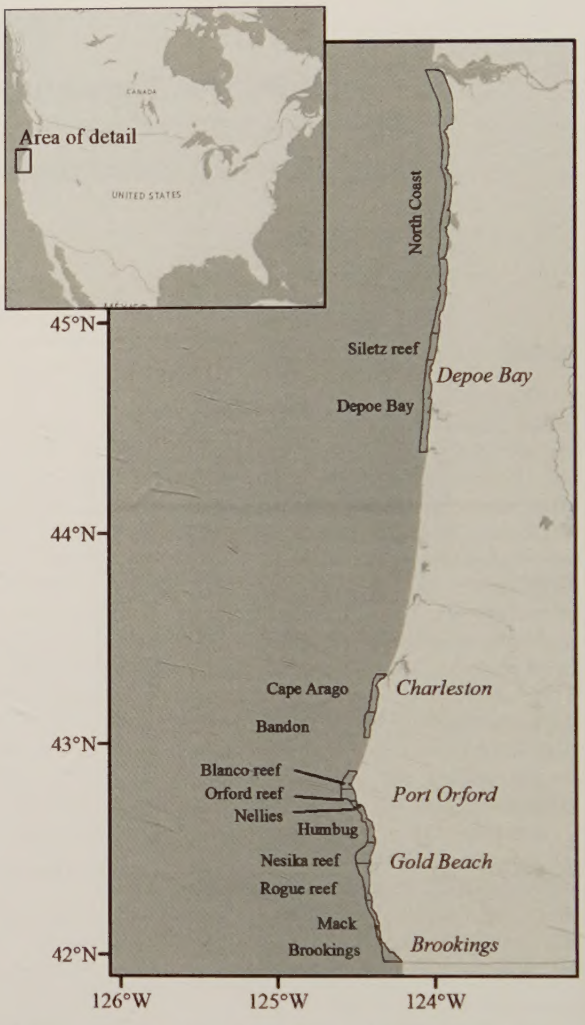


Figure 2.—Sea urchin "ODFW areas" (defined by continuous reef) in relation to ports important to sea urchin deliveries along the Oregon coast.

etc.) were developed around this budding industry. The stock was fished down; efficiency and then effort both declined. By 1996, the last remaining Oregon sea urchin processing facility closed (Richmond et al., 1997), and since that time, sea urchins have been trucked to adjacent states for processing. Since the late 1990's catch and effort has remained steady, albeit at a lower level.

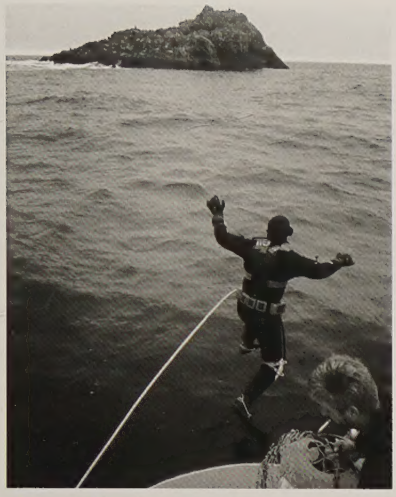
Fishery Scope

Oregon's red sea urchin fishery is an important component of the states' fishery profile with connections to the broader region. California has the most extensive kelp beds (thus fishing area) and consequently is the largest and most valuable component of the U.S. west coast fishery cumulatively totaling, 387,412 t from 1971 to 2022. Washington State has limited coastal kelp beds but extensive kelp beds within Puget Sound. Landings in Washington have totaled 23,366 t between 1981 and 2022. Oregon's fishery is similarly sized to Washington, totaling 20,587 t from 1986 to 2022 (Pacific States Marine Fisheries Commission, 2022). Overall, the U.S. west coast red sea urchin fishery had a strong peak centered around 1990, when factors such as strong markets and exploration of previously unexploited stocks caused the fishery to boom (Fig. 1).

Oregon's coastline measures 362 miles (636 km) and is characterized by rocky shores in southern portions, while the northern portions of the state tend to be sandy. Given the predominance of rocky shorelines in the southern portion of Oregon, kelp beds (mostly *Nereocystis luetkeana*) occur most prominently. Importantly, two large offshore rocky reefs account for most of Oregon's kelp beds: Orford Reef and Rogue Reef hold most of Oregon's kelp beds (Merems and Donnellan, 2011), and consequently the majority of the red sea urchin catch (71%) has occurred at those areas (Fig. 2).

Purple and Green Sea Urchins

The Oregon sea urchin fishery focuses on red sea urchin; however, the purple sea urchin, *Strongylocentrotus purpuratus,* is a minor component of landings, accounting for less than 1% of historic catch (Pacific States Marine Fisheries Commission, 2022). Green sea urchin, *Strongylocentrotus*



Sea urchin diver jumping in the water while tender prepares gear. Photo: Scott Groth.

droebachiensis, a target of fisheries north of Oregon (Washington, Canada, and Alaska) are uncommon in Oregon; however, they have been found to be present in extremely small numbers for the first time in 2015, near Port Orford (personal observ.).

Methods and Materials

Fishery Methods

The fishery for red sea urchin is artisanal in nature, requiring specialized diving gear and a high degree of skill to take quality products. Divers use a surface supplied air system, referred to as "hookah," which allows them an unlimited supply of air. Divers pick individual sea urchins off the reef using a hand rake and place them in suspended mesh bags, made neutrally buoyant with the aid of a buoy of lift bag system. This enables divers to move their catch and mesh bag along as they swim the reef. Since sea urchin are individually collected, there is no bycatch.

Dive tenders (non-diving crewmembers) keep lines clear, assist getting divers and catch on and off the vessel, and assure that things are running smoothly while divers are underwater. Most sea urchin divers begin their careers as dive tenders to gain experience on the water because commercial diving requires more skill and knowledge compared to most other fisheries and has a higher degree of danger.

Sea urchin vessels make single-day trips, returning to port each day for processing. Vessels in the fleet are mostly small boats (<35 ft long) with a shallow draft for fishing tight and shallow areas. During a single day trip, each diver (usually 1–4 individuals) makes multiple dives, constrained by sunlight and safe diving time. Live catch is returned to port daily and then trucked to remote processing facilities. Processing requires cracking the sea urchin's test and removing the five skeins of gonad (erroneously called "roe") (Kato and Schroeter, 1985). Less frequently they are sold whole in live markets. The quality and quantity of sea urchin gonads in each landing determines the value, not strictly the whole wet weight landed at port as in other fisheries.

Catch, Effort, and Stock Metrics

Landing receipts and fishery logbooks for 1986–2022 were used to assess catch and effort. Buyers fill out a receipt for each landing of sea urchins and must record diver name, date, port landed, total pounds for each species, and the unit price. To assess catch by reef, logbooks from 1986 to 2022 were used. Logbooks require the recording of catch area (e.g., reef name), depth, diving time, and estimated weight of catch. From 1986 to 2011 divers recorded logbook information as a daily summary, consisting of a coarse description of area and estimated weight of catch. Beginning in 2012 divers recorded logbook information for individual dives, including GPS data combined with estimated catch weight from each dive. To measure stock metrics, we used market sampling and population surveys.

Market Sampling

To understand changes in the size distribution of red sea urchin caught in the fishery, market sampling was employed. During opportunistically selected deliveries, biologists measured the TD of 50 randomly selected red sea urchin using 300 mm Vernier calipers. Data were pooled according to catch area and by year, (Fig. 2 shows the areas). Occasionally, divers targeted only the largest red sea urchins to satisfy specific markets; those deliveries were not sampled. Market sampling at Orford Reef was robust and continuous, while other areas were fished and sampled less frequently (Table 1).

Population Surveys

Index sites were used to measure the abundance and size distribution of red sea urchin populations in areas critical to the fishery. Survey areas were selected based on three criteria: 1) relevance to the fishery (e.g., prime catch areas, reserve areas, etc.), 2) expected presence of commercial quantities of red sea urchin, and 3) depths were generally shallower than 20 m. Surveys were performed periodically from the early 1990's to 2019, though earlier survey data by Oregon State University (OSU) from 1983 and 1984 was also incorporated (Washburn, 1984). Red sea urchin population surveys were conducted at three ports (Port Orford, Depoe Bay, and Charleston) important to the fishery, and, in each, fished and reserve areas were surveyed over time (Table 2).

Surveys were performed using subtidal belt transects. At each transect, two divers worked together to extend a 30 or 40 m line, then identify and count sea urchin species within 1 m perpendicular to each side of the transect line (Fig. 3). Survey methodology focused on emergent sea urchins, rocks were not moved, and lighting was not used.

For size distribution, two methods were used. For surveys from the 1990's, the first 50 red sea urchins in each transect were measured in situ, when fewer than 50 were found, those areas adjacent to the transect were searched until 50 were measured. For surveys from the 2010's, all sea urchins were collected and brought to the surface for measurement, when sea urchins were too numerous for this to

Sea urchin diver with a deckload of red sea urchins. Photo: Scott Groth.

be practical the first 50 of each sea urchin species were collected, when less than 50 sea urchins (per species) were found, no extra efforts to measure sea urchins at the site were made. Importantly, population survey siting focused on the red sea urchin, not the purple sea urchin, which is found at highest densities shallower than these surveys occur.

Survey Areas

Three ports were selected to index red sea urchin populations in Oregon: Port Orford, Depoe Bay, and Charleston. Selection was based on importance to the fishery, accessibility, and geographical separation. Transect sites were repeated as geographically exactly as possible in each sampling event. The number of sites surveyed at each area in a year was constrained by budget.

Table 1.—Mean test diameter size (mm) and number of red sea urchin from market samples, by ODFW area, 1987–2022.

	Rogue		Humbug		Orford		Arago		Depoe			
Year	*n*	Mean size (mm)	*n*	Mean size (mm)	*n*	Mean size (mm)	*n*	Mean size (mm)	*n*	Mean size (mm)	Total *n*	Mean size (mm)
1987					50	131.40					50	131.40
1988	30	120.60	55	120.96	763	108.05	125	120.45			973	110.76
1989	678	128.60	86	107.65	2,381	124.05	150	122.89	33	113.64	3,328	124.40
1990	1,761	126.83	2,190	120.79	7,420	128.34	350	123.81	367	130.41	12,088	126.68
1991	975	122.56	150	106.43	2,570	116.20	75	111.77	330	113.53	4,400	117.49
1992	150	119.66	250	113.16	2,567	113.93	125	114.81	350	118.97	3,492	114.86
1993					1,205	110.53	474	110.88	1,149	105.12	2,828	108.39
1994	100	134.85	100	110.67	1,656	111.83	50	114.92	290	110.46	2,196	112.71
1995	200	114.25			600	112.11					800	112.65
1996	450	110.32			600	108.36			50	101.12	1,100	108.83
1997	50	106.82			950	106.81					1,000	106.81
1999					1,052	106.73					1,052	106.73
2000					1,600	107.49					1,600	107.49
2001					1,248	106.74					1,248	106.74
2002					150	104.67					150	104.67
2004					149	110.14					149	110.14
2005			107	116.03	320	117.14	243	114.05			670	115.84
2006	207	123.08	50	105.32	203	122.81					460	121.03
2007					183	116.15					183	116.15
2008					634	113.54			200	115.61	834	114.04
2009					1,592	111.54					1,592	111.54
2010			250	118.14	896	120.27	309	119.58			1,455	119.76
2011			50	120.21	3,029	120.27					3,079	120.27
2012					499	117.23	872	115.92	500	115.03	1,871	116.03
2013					1,100	119.83					1,100	119.83
2014					900	126.24					900	126.24
2015					550	128.05					550	128.05
2016					100	127.93					100	127.93
2017	100	100.46	50	116.82	399	109.24					549	108.33
2018	142	95.56									142	95.56
2019	54	94.21									54	94.21
Total	4,897		3,338		35,366		2,773		3,269		49,993	
Average		121.93		118.28		117.78		116.70		112.95		117.89

ODFW biologist measuring red sea urchin in situ during fishery independent survey. Photo: ODFW Archives.

Port Orford

Three areas were selected to index red sea urchin populations near Port Orford on Oregon's south coast (Fig. 4). Two fished areas were surveyed: 1) Orford Reef, a 1,367 hectare (ha) area 10 km northwest of the port of Port Orford and 2) Humbug Mountain, a 200 ha area 9 km south of Port Orford. One reserve area was also surveyed, Redfish Rocks Marine Reserve (MR), a no-take marine reserve, 5 km south of Port Orford. Prior to its establishment as a marine reserve in 2012 the Redfish Rocks area was an important sea urchin fishing area.

Depoe Bay

Five areas were selected to index red sea urchin populations near Depoe Bay, on Oregon's central coast (Fig. 5). Three fished areas were surveyed: 1) Government Point, a 160 ha area 1.6 km northwest of Depoe Bay, 2) Depoe Bay, a 240 ha area adjacent to Depoe Bay, and 3) Cape Foulweather, a 200 ha area south of Depoe Bay. Pirates Cove Research Reserve (RR) is a small (3 ha) no-take research reserve 1.1 km north of Depoe Bay, established

Table 2.—Summary of red and purple sea urchin densities (per m^2) by year and reef in Oregon, 1991–2019.

Port	Reef	Year	*n*	Mean red sea urchin density	Mean purple sea urchin density
Charleston	Gregory Point	1996	2	1.40	0.70
		1997	1	1.20	1.60
		2013	6	0.48	0.21
		2015	6	0.45	4.02
	Lighthouse	2013	5	1.24	0.00
		2015	5	1.47	1.00
	Simpson Reef	1993	1	0.20	0.00
		1996	1	0.20	0.00
		1997	1	0.40	0.00
		1999	1	0.60	0.00
		2013	18	0.19	0.01
		2015	17	0.22	0.11
Depoe Bay	Cape Foulweather	1991	2	0.08	0.00
		1994	10	0.67	0.01
		1996	10	1.11	0.05
		1998	9	0.58	0.00
		2012	11	0.27	0.01
		2015	12	0.21	0.23
	Depoe Bay	1991	4	0.85	0.10
		1994	6	1.62	0.62
		1996	6	2.62	0.02
		1998	3	1.97	0.00
		2012	7	1.49	1.13
		2015	7	0.72	0.48
	Government Point	1991	2	0.36	0.00
		1994	9	1.77	0.77
		1996	5	2.83	0.69
		1998	6	1.92	0.43
		2012	5	0.90	0.01
		2015	4	1.17	0.00
	Pirates Cove	2012	4	0.26	0.00
		2015	4	0.20	0.05
	Whale Cove	1996	1	0.53	0.01
		1997	1	0.80	0.00
		1998	1	1.80	0.00
		2012	6	0.42	0.00
		2015	6	0.50	0.01
Port Orford	Humbug	1992	5	0.41	0.01
		2011	15	0.09	0.00
		2014	9	0.13	0.00
		2016	17	0.29	0.53
		2019	9	1.54	3.89
	Island Rock	1984	1	1.70	0.00
		1992	4	0.73	0.01
	Nellies Cove	1984	4	3.07	0.00
		1992	3	0.88	0.04
		1993	1	0.96	0.01
		2015	19	1.52	2.57
		2019	11	2.02	14.93
	Orford Reef	1984	6	2.71	0.00
		1991	37	0.88	0.00
		1993	39	0.90	0.05
		1995	41	0.68	0.08
		1997	15	0.65	0.07
		2011	39	0.27	0.00
		2014	36	0.52	0.06
		2016	30	3.29	2.52
		2019	24	4.66	6.24
	Redfish Rocks	1984	4	2.60	0.00
		1992	2	2.28	0.03
		2011	16	0.42	0.00
		2014	8	0.74	0.00
		2016	15	1.13	0.28
		2019	9	1.87	0.66

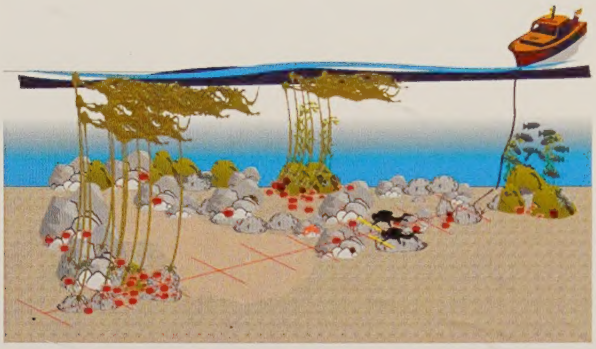


Figure 3.—Illustration of methodology for subtidal belt transects used in ODFW sea urchin population surveys (illustration by Scott Groth).

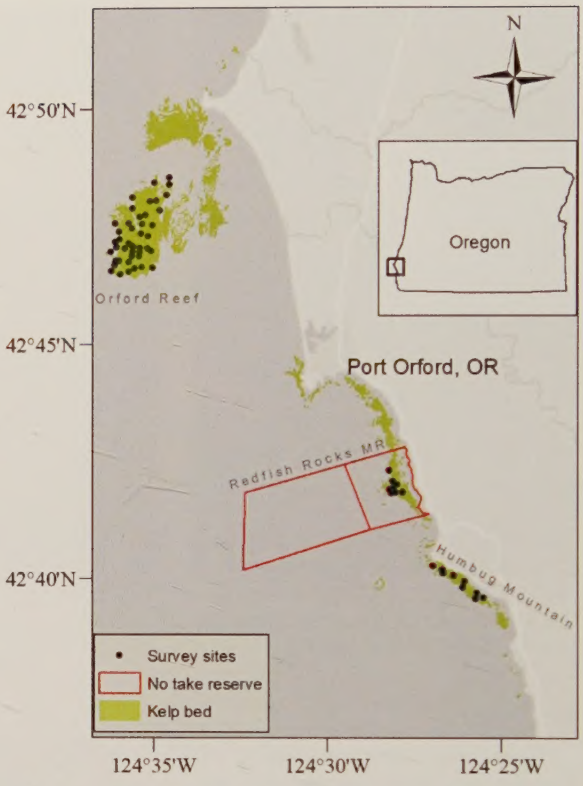


Figure 4.—Sea urchin population index survey sites near Port Orford.

in 1999. Whale Cove Habitat Reserve (HR) is a small (13 ha) no-take reserve established in 1963, 2.5 km south of Depoe Bay. Otter Rock MR is a small no-take marine reserve, 7 km south of Depoe Bay established in 2012; however, data from this area was combined with Cape Foulweather since its adoption was recent and long-term fishing pressure has been minimal.

Charleston

Three areas were selected to index red sea urchin populations near Charleston, a port on Oregon's south-central coast (Fig. 6): 1) Simpson Reef, an expansive fishing area (300 ha) surrounding Cape Arago and 7 km southwest of Charleston; 2) Gregory Point RR, a 24 ha no-take reserve, established in 1993 and 4 km west of Charleston; and 3) Lighthouse Beach, a small (13 ha) but geographically separate area 13 km south of Charleston and adjacent to the north edge of Gregory Point RR.

Results

Catch, Effort, and Value

Red sea urchin fishery landings in Oregon developed quickly (1986–88), expanded and then decreased dramatically (1989–1995). Finally, they were reduced to a stable level from 1996 to 2022. The fishery began in 1986, and by 1990, 4,228 t were landed. By 1996, landings had dropped dramatically, averaging 217 t (annually) from 1996 to 2022. Effort (measured number of individual trips) closely mirrors total landings, with some lag as the stocks were depleted, going from 31 in 1986 to 4,435 in 1991. In the most recent 10-yr period (2013–22), effort has been stable, averaging 325 trips per year (Fig. 7).

Similar to catch and effort, the value of the fishery experienced rapid expansion, then became stable at a lower level. Value of the fishery peaked in 1990 with an ex-vessel value of $3.4 million in 1990; at that time, markets were excellent and Oregon was able to develop processing infrastructure, further elevating ex-vessel value. Fishery stock and markets quickly declined, and the fishery became smaller but stable. From 1996 to 2017, value averaged $284,000 per year. Recent years (2018 to 2022) have seen highly increased prices and values consequent with low kelp conditions along the U.S. west coast. Fishery value from 2018 to 2022 averaged $673,400 annually.

Price per pound of red sea urchin has been variable, with recent sudden changes (Fig. 8). Mean price per pound in the early years was high (e.g., $1.93/kg in 1993) as markets were lucrative. At that time, the Japanese economy (the primary market) was booming and previously unexploited U.S. west coast stocks provided a robust source. As the stock was depleted and the Japanese economy turned down, prices declined. As the fishery matured and stock levels remained low, domestic markets were used more frequently. Beginning around 2016, U.S. west coast kelp beds decreased, reducing the amount of high quality uni, leaving just a few areas of Oregon still viable for markets causing the price to elevate, and from 2018 to

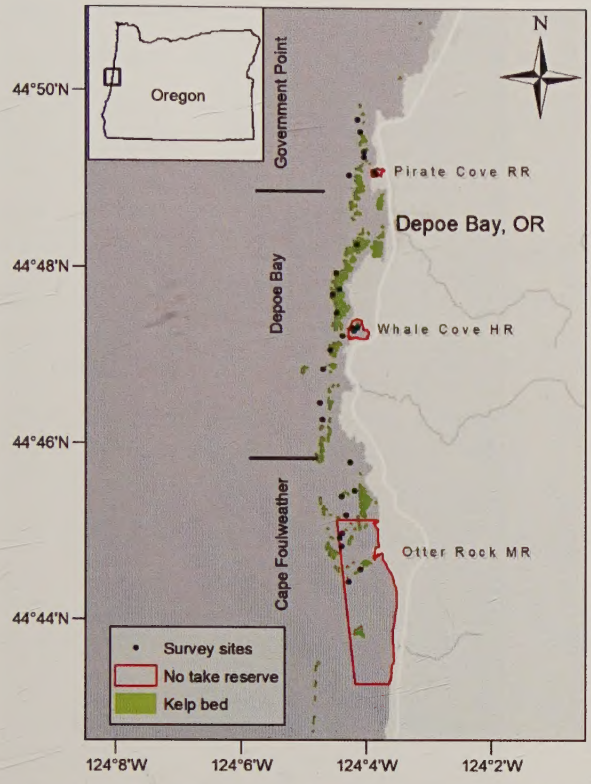


Figure 5.—Sea urchin population index survey sites near Depoe Bay.

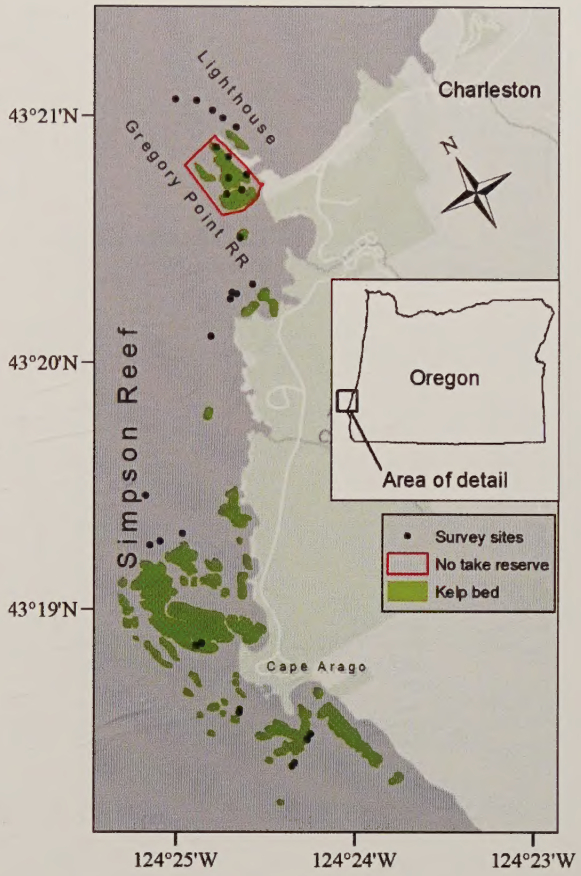


Figure 6.—Sea urchin population index survey sites near Charleston.

2022 prices were at all-time highs (averaging $6.10/kg).

Catch by Port

Oregon's red sea urchin fishery catch is focused on the southern portion of the state. In total 20,587 t have been caught across the 1986–2022 period. By port, Port Orford has had the highest landings (11,733 t) followed distantly by Gold Beach (4,779 t), then Depoe Bay (1,820 t), Charleston (979 t), Brookings (724 t), and Newport (517 t), while other ports including Garibaldi and Pacific City (combined) added 35 t (Table 3).

Catch by Fishing Area

Most of the Oregon red sea urchin landings originate from the south coast at Orford Reef and Rogue Reef. On average, these two areas account for 71% of the landings from years 1986 through 2022 (Fig. 9). Fishery logbooks also validate this fact as diver-based accounts show that most landings are occurring here, and that they are strongly associated with the presence of surface canopy kelp (Fig. 10). This strong correlation has not been evaluated thoroughly, but future analysis may include modeling or statistical analyses showing that kelp canopy can be a strong predictor of sea urchin catch.

Catch by Depth

Time at depth constrains fishing (dive) time in this fishery. As a diver descends, the amount of time one may stay underwater declines, due to increased nitrogen absorption. Typically, sea urchin divers attempt to stay as shallow as possible (for increased dive time), while considering sea state conditions (swell, current, etc.). The mean diving depth of fishing was 14.6 m, deeper at offshore reefs (Orford Reef (16.7 m), Rogue Reef (14.8 m)), and shallower at nearshore areas (Depoe Bay (9.3 m), Charleston (8.7 m), and Brookings (10.4 m)).

Stock Metrics

Market Sampling

Catch at Orford Reef shows a pattern familiar to the "boom and bust" of a virgin stock. Initial market sampling showed catch of primarily large, older animals in the first years of the fishery (1986–90). From 1991 to 2002, a period of "recruitment fishing" occurred, where catch was dominated by red sea

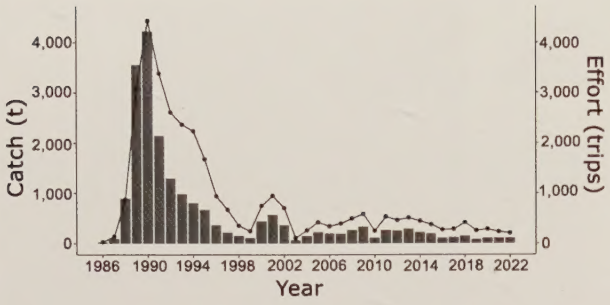


Figure 7.—Catch (t, bar chart) and effort (number of trips, line chart) in Oregon's commercial red sea urchin fishery, 1986–2022.

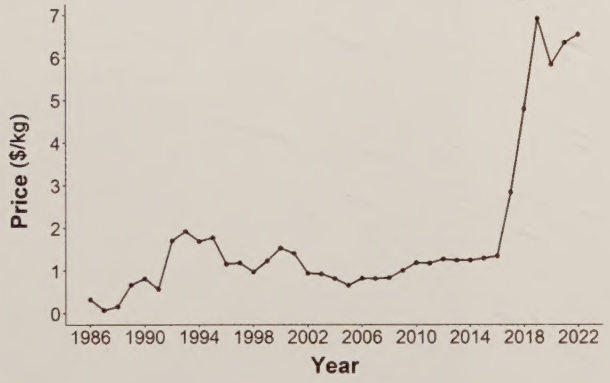


Figure 8.—Nominal price $/kg for red sea urchin in Oregon, 1986–2022.

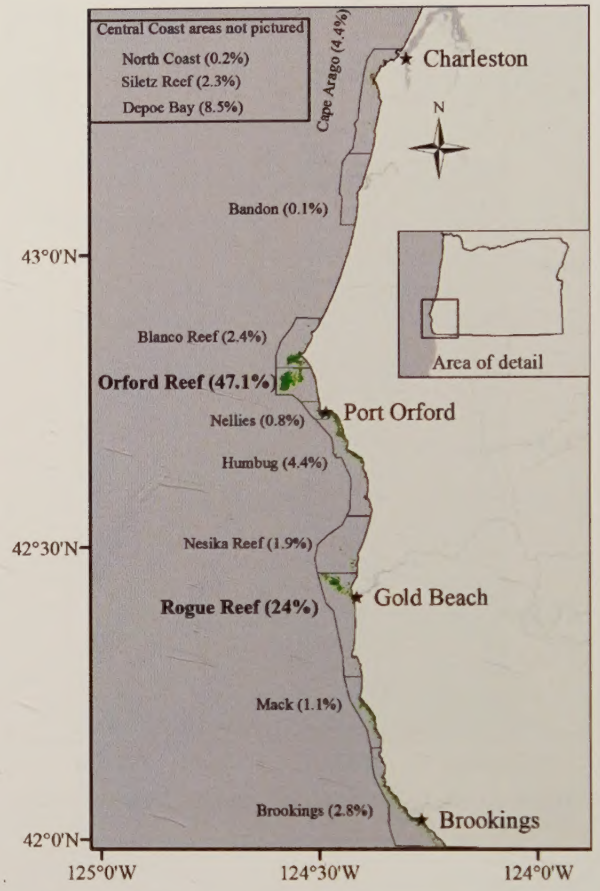


Figure 9.—Catch of red sea urchin by ODFW area, in Oregon, 1986-2022. Note emphasis on south coast areas particularly, Orford Reef and Rogue Reef.

urchin, simply recruiting to minimum legal size (MLS). From 2004 to 2016 when red sea urchin fishing pressure was lower, larger and older individuals again became a substantial component of catch. Sampling during 2017–18 shows smaller individuals in the catch again, probably due to incoming cohorts of fast-growing red sea urchin from recent episodic events recruiting to MLS (Fig. 11).

The early years of market sampling show many large and old red sea urchin caught at Orford Reef. Several individuals measuring >180 mm TD were caught in the first years of the fishery, including a maximum size of 197 mm found in a 1989 market sample. No red sea urchin of this size has been found after this initial fish-down of the stock; however, they have been found at this size in population survey samples of Whale Cove HR, the only area of the state established as a no-take reserve prior to the inception of the fishery. This is evidence of the intensity of the fishery and the long lives of the red sea urchin.

Population Surveys

Together, abundance (i.e., number of sea urchins per m^2) and size distribution (i.e., sizes (expressed in TD)) provide insight into the population dynamics of the red sea urchin stock.

Port Orford The areas of Port Orford are most important to Oregon's red sea urchin fishery. Population surveys have focused primarily on Orford Reef, where a majority of Oregon's kelp bed area and fishery catch occur. Prior to the beginning of the fishery, stocks were robust but then were quickly fished down, to the extent that fishing interest dropped sharply. In recent years densities abruptly increased to pre-fishing levels.

Orford Reef Given its extent of shallow rocky grounds and kelp beds, Orford Reef is the highest value fishing area. Hence, it is a critical area to understand red sea urchin population abundance, size distribution, and recruitment in Oregon. Prior to fishery inception, red sea urchin densities were high (2.71/m^2 in 1984) and composed of large, old individuals (Washburn, 1984). From 1988 to 1997, fishery effort and landings were high. During this period, densities of red sea urchin were substantially reduced, and

Table 3.—Landed weight (mt) and nominal value ($) for red sea urchin in Oregon, 1986–2022.

	Port								Totals	
Year	Brooking	Gold Beach	Port Orford	Charleston	Newport	Depoe Bay	Garibaldi	Pacific City	Landings (t)	Value ($)
1986	0	0	25	0	0	0	0	0	25	8,313
1987	22	0	70	0	0	0	0	0	92	7,025
1988	24	102	681	81	1	4	0	0	894	141,731
1989	29	719	2,736	29	40	4	0	0	3,557	2,377,331
1990	52	1,174	2,230	132	100	523	9	9	4,228	3,447,461
1991	154	545	1,079	146	0	220	0	4	2,149	1,225,653
1992	65	174	769	39	25	224	0	0	1,296	2,212,784
1993	58	118	499	67	175	74	0	0	990	1,908,848
1994	26	229	480	16	6	55	0	0	812	1,372,999
1995	59	223	316	9	20	51	0	0	678	1,205,142
1996	15	136	202	11	4	3	0	0	372	431,388
1997	17	49	137	0	1	18	0	0	222	263,634
1998	21	56	49	20	2	8	0	0	156	152,164
1999	14	27	52	5	7	7	0	0	113	138,851
2000	3	123	168	85	9	58	0	0	446	682,484
2001	62	5	155	47	126	173	2	0	571	803,287
2002	62	42	97	55	0	112	0	0	368	347,879
2003	1	0	34	16	0	14	0	0	65	60,501
2004	0	3	69	41	0	37	0	0	151	122,670
2005	0	6	127	70	1	18	2	0	224	147,883
2006	0	48	91	12	0	50	2	0	203	166,185
2007	0	52	115	28	0	0	0	0	195	159,404
2008	0	13	182	20	0	51	2	0	268	223,168
2009	0	111	224	2	0	0	3	0	341	341,907
2010	5	15	93	2	0	0	0	0	114	134,711
2011	3	116	140	1	0	7	0	0	267	313,488
2012	1	29	131	12	0	85	0	0	258	327,982
2013	0	59	225	2	0	10	0	0	296	367,116
2014	0	25	204	0	0	0	0	0	229	284,508
2015	9	20	164	4	0	6	0	0	202	260,307
2016	0	9	102	0	0	0	0	0	112	149,518
2017	21	28	56	16	0	7	0	0	128	362,362
2018	0	138	13	0	0	0	0	0	151	725,445
2019	0	61	16	5	0	0	0	0	82	570,331
2020	0	105	1	5	0	0	0	0	111	652,044
2021	0	111	0	0	0	0	0	0	111	702,791
2022	0	109	0	0	0	0	0	0	109	716,266
Total	724	4,779	11,733	979	517	1,820	21	14	20,587	23,515,560

by 1997 densities were low (0.65/m^2), when compared to "commercial quantities" (conventionally, one red sea urchin/m^2). Across this period, effort and landings dropped sharply (Fig. 7). By 2011, after years of lower effort and landings, stocks had still not rebounded and were at their lowest levels found in this study. Strong recruitment events occurred throughout the mid-2010's, driving densities to their highest level (4.66/m^2 in 2019), fully recovering from the long and exploitive fishing effort (Fig. 12a).

In 1991, ODFW performed the first robust red sea urchin population survey of Orford Reef. Red sea urchin populations were dominated by very old, large individuals, combined with many smaller individuals, likely from a recruitment event in the late 1980's. The standing stock of old individuals was quickly removed (by the fishery) and the population was dominated by recruits from the 1980's episodic recruitment event until the mid-2010's. First detected in 2014, another strong episodic recruitment event occurred and has become the primary component of the recent stock (Fig. 12b).

Purple sea urchin populations at Orford Reef were nearly absent (in the deeper waters of these surveys), but they increased suddenly, beginning in 2014. Few purple sea urchins were found in early surveys (e.g., <0.01/m^2 in 1991), when suddenly in 2016, high abundances were found (2.52/m^2), then increasing further in the most recent 2019 surveys (6.24/m^2) (Fig. 13). The size of individual purple sea urchins at Orford Reef is generally small, with mean size at about 43.7 mm in 2019 (Fig. 14).

Humbug Mountain The rocky reefs adjacent to Humbug Mountain are less expansive than at Orford Reef, though they are geographically separated, allowing a good comparison area. The first surveys were conducted on Humbug Mountain in 1992. At that time, the fishery had already operated at this area for several years. Survey results from 1992 showed substantial densities (0.41/m^2) of red sea urchin and few purple sea urchin (0.01/m^2). Like Orford Reef, Humbug Mountain was fished down substantially by 2011. By 2019, red sea urchin populations recovered beyond the virgin stock conditions (Fig. 14a). Size distribution also showed a similar pattern as Orford Reef, where 1992 sur-

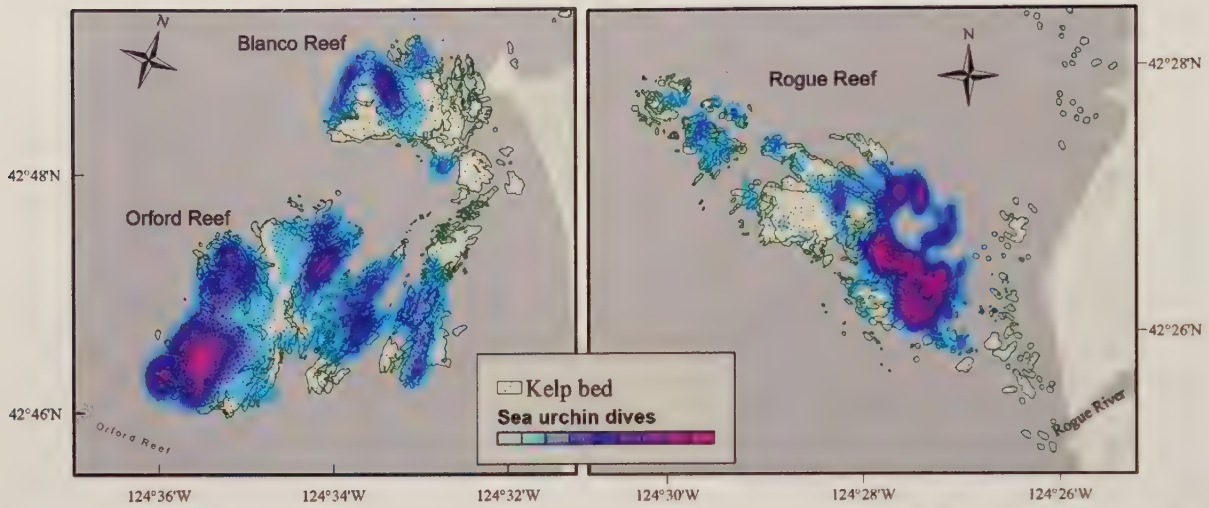


Figure 10.—Heat map of red sea urchin dives in relation to kelp bed extents (from Merems and Donnellan, 2011) at Orford Reef and Rogue Reef, 1986–2022.

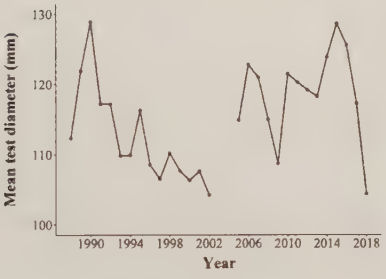


Figure 11.—Mean test diameter (mm) of red sea urchin caught in the commercial fishery at Orford Reef, 1986–2018 (no sampling since 2018).

veys showed dominance by a ~90 mm TD size class, then a large (~135 mm TD), old population in the mid-2010's followed by a late 2010 large recruitment event, where the population became dominated by ~40 mm red sea urchin (Fig. 14b).

Redfish Rocks MR The rocky reefs of Redfish Rocks MR were designated a no-take reserve in 2012; kelp beds in the rocky nearshore area are abundant. Red sea urchin surveys have occurred at regular intervals in the 2010's, but only two historic surveys (1984 and 1992) exist. Although early surveys show a high density of red sea urchin (e.g., 2.28/m^2 in 1991), by 2011 densities were lower (0.42/m^2); however, since that time, recruitment there has been consistent, and densities have increased steadily (Fig. 14c). Like Orford Reef and Humbug Mountain, size distribution at Redfish Rocks MR show that 1992 populations were dominated by a ~90 mm TD size class, then a large, old population in the mid-2010's, followed by a late 2010's large recruitment event, where the population became dominated by ~40 mm individuals (Fig. 14d).

Humbug Mountain and Redfish Rocks have both experienced dramatic increases in purple sea urchin densities which were low from 1992 to 2014 (0.01 and 0.03/m^2, respectively). Then densities became very high by 2019 (3.89 and 1.87/m^2, respectively) (Fig. 14a, c).

Depoe Bay The Port of Depoe Bay is located on Oregon's central coast, and it accounts for 9% of Oregon's red sea urchin fishery catch; kelp beds are small and disconnected from primary area in the south coast. Survey areas encompassed nearly all the ports' kelp beds and included three no-take reserve areas.

Red sea urchin populations within the fished areas of Government Point, Depoe Bay, and Cape Foulweather are situated north to south, respectively. Throughout the areas, the larger and older individuals were removed from the population in the early years of the fishery; afterward, a strong recruit class settled in this region. Red sea urchin populations at all fished areas near the Port of Depoe Bay were dominated by a single year class which likely settled around 1992. Anecdotally, kelp beds within this region were more persistent in the southern areas than those in the north, and this appears to reflect the growth of red sea urchins which has been faster in the south than the north but is also coupled with greater densities in northern areas (Fig. 15b, d, f).

At Government Point and Depoe Bay, population densities of red sea urchin have been high. Densities peaked in 1996 at 2.83 and 2.62/m^2, respectively, driven by the single recruit class of ~1992 (Fig. 15a, c). Despite high populations, individual sizes have just barely reached the fishery MLS of 89 mm TD, despite at least 26 yr to grow. It seems likely their very slow growth could be caused by low amounts of kelp and high densities of red sea urchin throughout the era. Densities have declined substantially through the years, probably through natural mortality with some contribution from fishery mortality.

At Cape Foulweather, red sea urchin densities have been persistent-

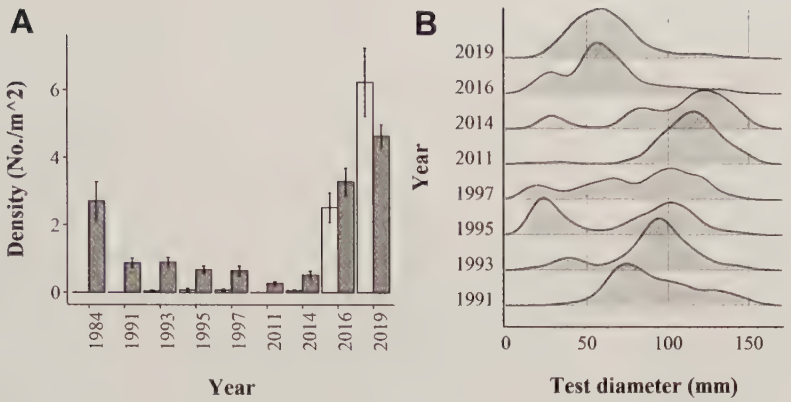


Figure 12.—Sea urchin stock dynamics at Orford Reef, 1984–2019, A) densities of red sea urchin (shaded bars) and purple sea urchin (white bars) by year (error bar indicates SE), and B) relative size distribution of red sea urchin by year, from survey data.

ly low (Fig. 15e); however, they have grown more quickly than other areas (Fig. 15f). The low densities combined with shallow depths and related higher kelp availability at these areas have likely contributed to the faster growth.

Depoe Bay features two longstanding marine reserves: Whale Cove HR (est. 1963), the only area in the state designated a reserve prior to the inception of the red sea urchin fishery and Pirates Cove RR (est. 1993). In addition, Otter Rocks MR (est. 2012) is a newer no-take reserve, since only a few sites were surveyed and the short time span since its designation, data from this area was pooled with previously described Cape Foulweather for this analysis.

Both Pirates Cove RR and Whale Cove HR had populations of large, old red sea urchins; however, densities at Whale Cove HR were higher and their mean sizes were the highest in the state (143.8 mm TD in 2015) (Fig. 16). In addition, the largest red sea urchin ever found in an Oregon survey was collected at Whale Cove HR in 2015, measuring 185.5 mm TD.

As of 2015, purple sea urchin densities were low near the Port of Depoe Bay. However, those surveys occurred just prior to the population booms

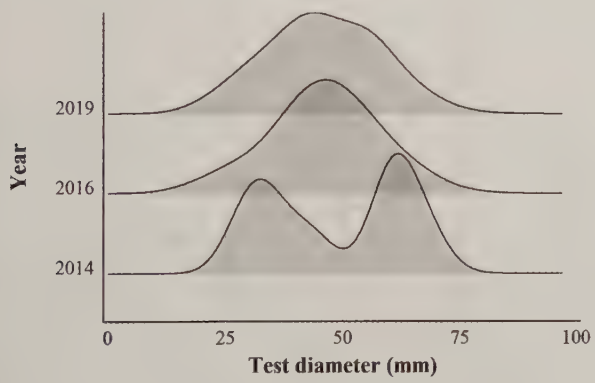


Figure 13.—Purple sea urchin size distribution at Orford Reef, 2014–2019, from survey data.

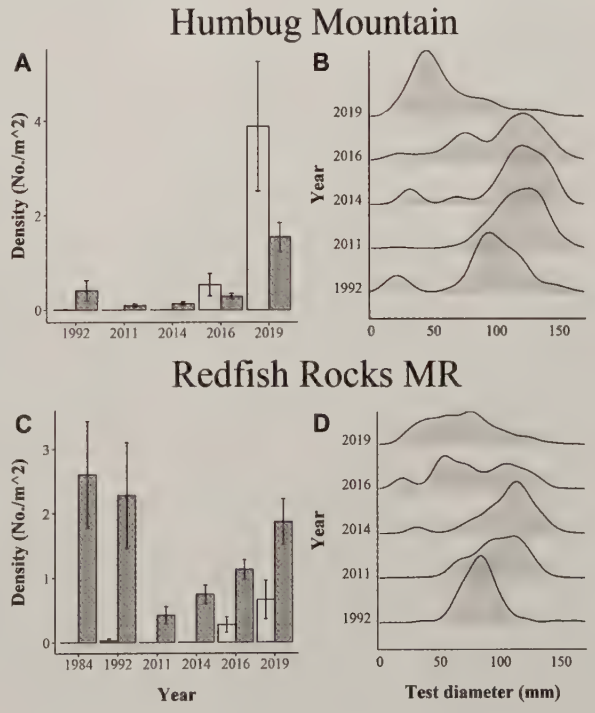


Figure 14.—Sea urchin stock dynamics at Humbug Mountain and Redfish Rocks Marine Reserve, 1984-2019. A and C) densities of red sea urchin, (shaded bars) and purple sea urchin (white bars) by year (error bar indicates SE), and B and D) relative size distribution of red sea urchin by year, from survey data.

found in 2016 and 2019 surveys of the Port Orford region.

Charleston The Port of Charleston accounts for 5% of Oregon's red sea urchin fishery catch. However, these populations are geographically separate from Port Orford and Depoe Bay and include a no-take reserve.

At Charleston, Simpson Reef is the primary sea urchin fishing area and the most expansive. Red sea urchin populations have been at a low level since surveys began (e.g., 0.20/m² in 1993) which was after initial, robust fishery removals. Little recruitment appears to have occurred at this area, and in the most recent surveys, densities were similar (e.g., 0.22/m² in 2015). Red sea urchin densities within Gregory Point RR have been higher than neighboring Simpson Reef, and mean sizes have been larger than other areas of Charleston (117.4 mm compared to 109.8 mm at Simpson Reef in 2015). Lighthouse Beach is a small area and is geographically separate from Simpson Reef and adjacent to Gregory Point RR. At Lighthouse Beach densities have been high recently (e.g., 1.47/m² in 2015); however, mean size has been smaller (e.g., 95.7 mm in 2015) and are presumably younger (Fig. 17).

As of 2015, the purple sea urchin boom found in southern ports just became noticeable in Charleston. While historic surveys showed low populations around Charleston (e.g., <0.01/m² in 1993 at Simpson Reef, 0.70/m² in 1996 at Gregory Point RR, and 0.20/m² in 2012 at Lighthouse Beach), many more were found in 2015 surveys. Purple sea urchin densities of 0.11/m², 4.02/m², and 1.00/m² at those same areas, respectively, were found in 2015 (Fig. 17a, c, e).

Management

Oregon's red sea urchin fishery management has focused on increasing sustainability, despite a massive boom then adjusting to a small fishery. Historical management actions are listed in Table 4. Key tenets of Oregon's red sea urchin fishery management include:

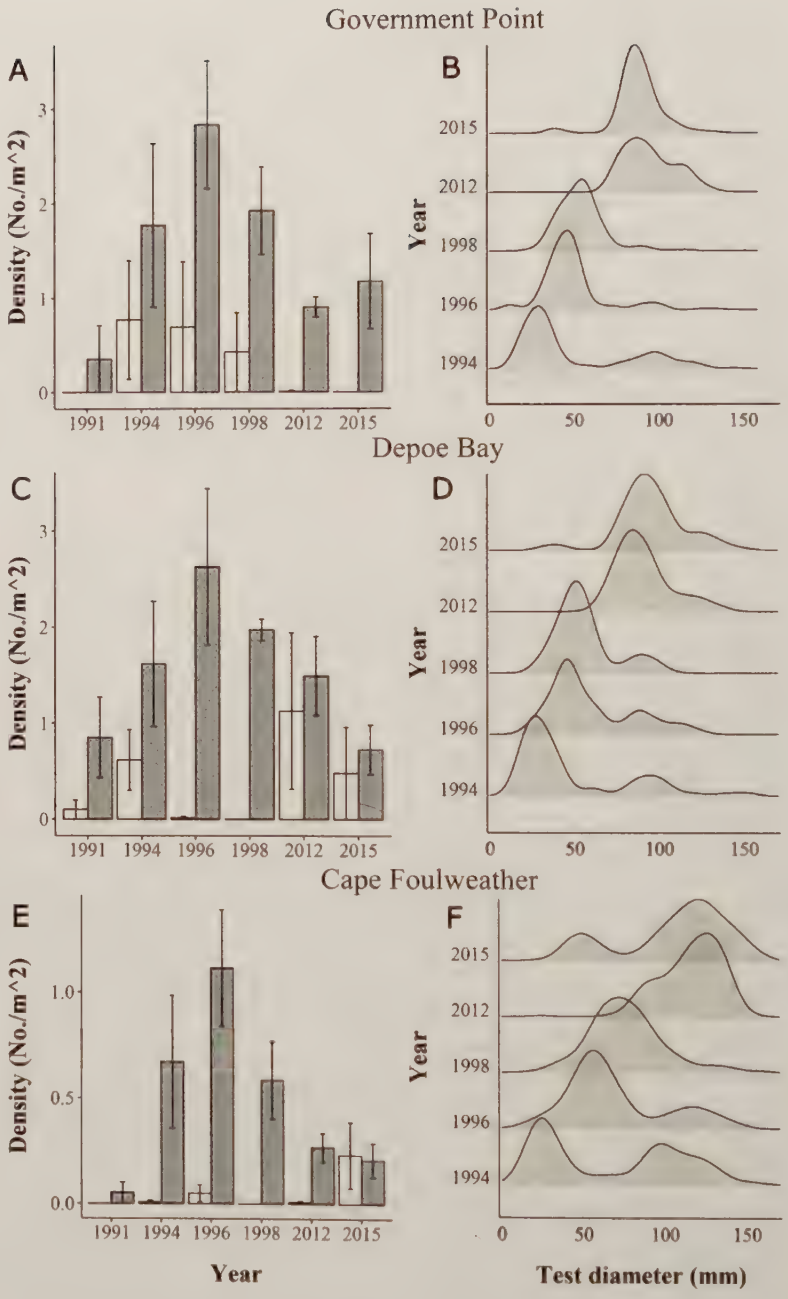


Figure 15.—Sea urchin stock dynamics at fished areas near Depoe Bay (Government Point, Depoe Bay, and Cape Foulweather) 1991–2015. A, C, and E) densities of red sea urchin (shaded bars) and purple sea urchin (white bars) by year (error bar indicates SE), and B, D, and F) relative size distribution of red sea urchin by year, from survey data.

1) Effort limitation: Limiting effort is a key to stabilizing fisheries. Limited entry allows managers to control pressure on the stock and reduces competition, increasing participant investment in sustainability. The sea urchin fishery boomed so quickly that limited entry was adopted soon after its inception (1987, 92 permits), to 46 permits in 1986, 30 in 1995, and 12 in 2016. The current level of 12 permits is designed to provide stability to the fishery and was reduced based on past levels of fishing that did not allow consistent fishery catch.

2) Area restrictions: Reserve areas, whether they be no-take reserves or de facto reserves, provide sites for the stock to persist without fishery pressure, enhancing source populations for the stock. Oregon red sea urchin fishery regulations include designated no-take reserves (e.g., Redfish Rocks RR, Whale Cove HR, etc.), which may be used as stock reserves and allow assessment of the stock absent fishery pressure.

Additionally, regulations creating de facto reserves may be easy to adopt. In Oregon's red sea urchin fishery, rules explicitly disallow fishing in very shallow zones (<3.3m water depth), but also practical limitations (e.g., depth and dive time) create additional de facto stock reserve in deep zones. In 2016, a regulation was adopted which disallows the use of mixed gas diving, increasing deepwater refugia.

3) Minimum legal size: Minimum size limits are used commonly in fisheries to protect immature stock and allow a period of reproductive viability prior to recruitment into the fishery. In Oregon's red sea urchin fishery, a minimum legal size was first adopted in 1988 (76.2 mm TD (3 inches)) then increased to 88.9 mm TD (3.5 inches) in 1991.

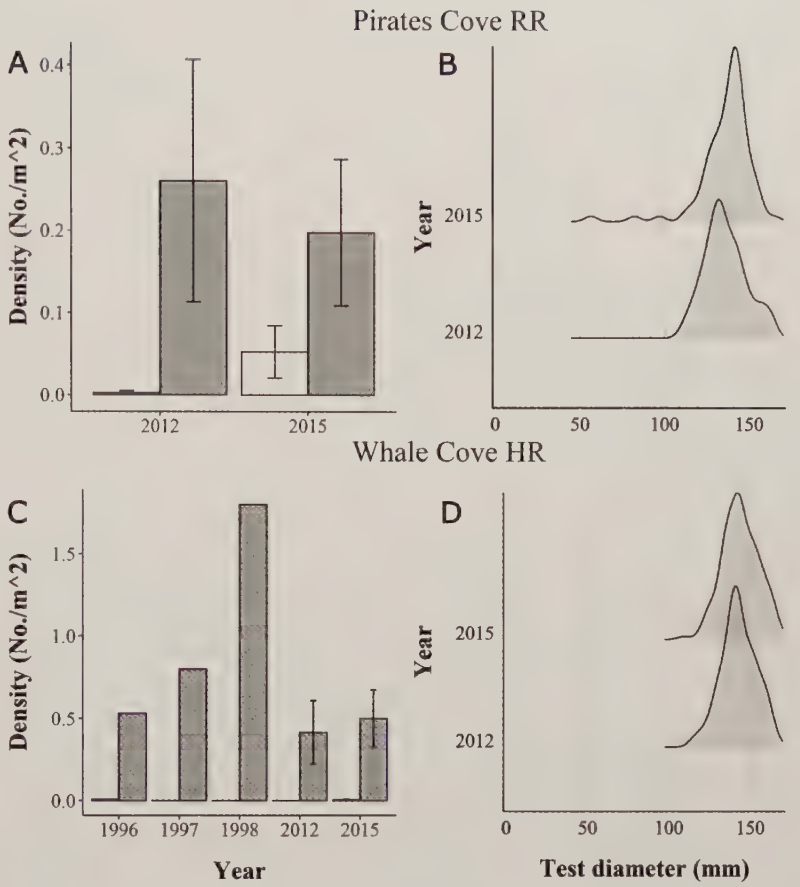


Figure 16.—Sea urchin stock dynamics at two subtidal reserve sites near Depoe Bay (Pirates Cove Research Reserve and Whale Cove Habitat Reserve) 1996-2015. A and C) densities of red sea urchin (shaded bars) and purple sea urchin (white bars) by year (error bar indicates SE), and B and D) relative size distribution of red sea urchin by year, from survey data.

Discussion

Stock Trends

Oregon's red sea urchin fishery exemplifies a "boom and bust" fishery, where a large virgin stock is fished out with only weak annual recruitment events. Soon after its inception the new fishery escalated, effort and landings rose quickly and then just as quickly fell (Fig. 7). Red sea urchin densities went from high to minimal in a few years (Fig. 12a). As the fishery progressed, it relied on the remnants of that very old population, and little recruitment input. By the early 2010's, the remaining stock was only large sizes in sparse populations (Fig. 12b). Recently however, densities have fully recovered from more than 30 years of fishing down.

Red sea urchin recruitment occurs episodically. During this study period (1983–2019), three large recruitment events have occurred in Oregon. A late 1980's recruitment event occurred throughout Oregon, which aided in fueling the robust fishery of the early 1990's. In the Depoe Bay region, an episodic event occurred around 1992 (Fig. 15). In 2014–16, another massive recruitment event was detected in the Port Orford region, though these red sea urchins have not reached fishery MLS (Fig. 12, 14). These events demonstrate their importance to the fishery where stocks may be severely depleted or overly abundant at times; however, it may not necessarily imply that the stock is over or underfished. It is clear that populations of sea urchins in Ore-

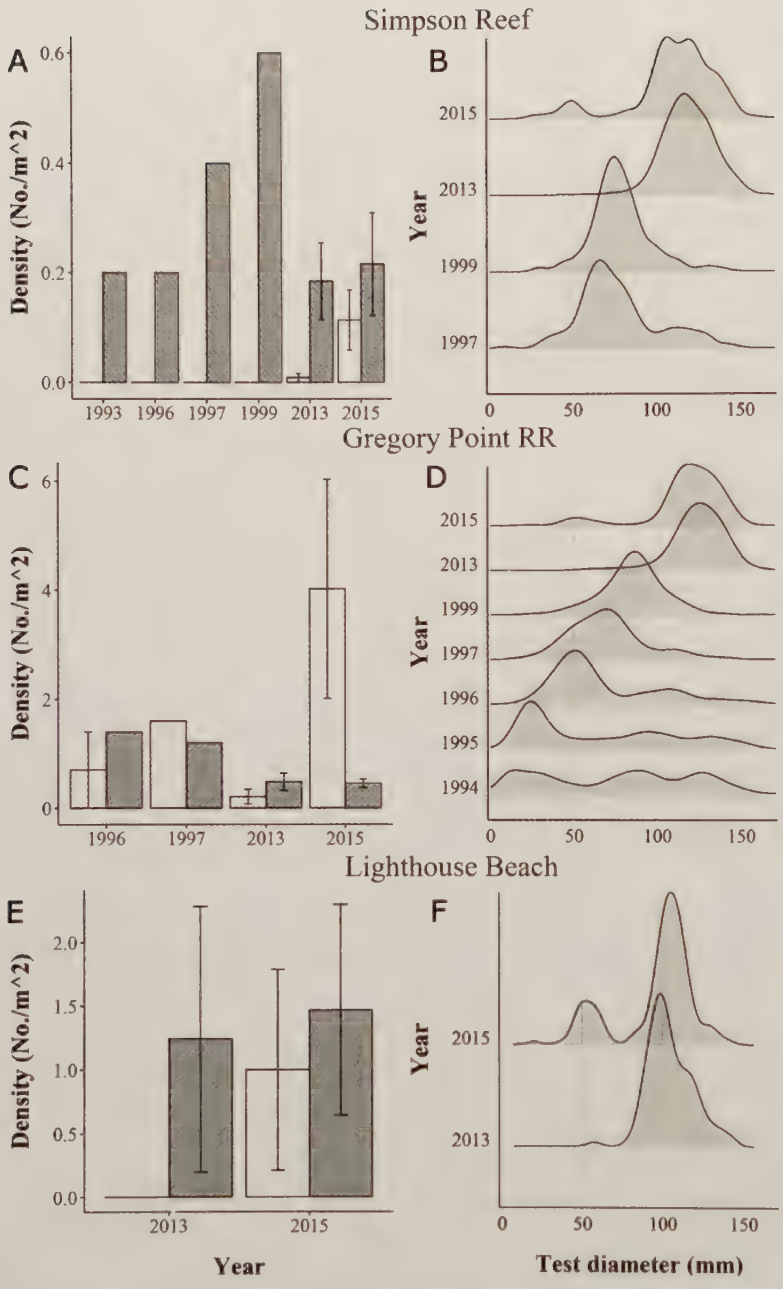


Figure 17.—Sea urchin stock dynamics at areas near Charleston (Simpson Reef, Gregory Point Research Reserve, and Lighthouse Beach) 1993–2015. A, C, and E) densities of red sea urchin (shaded bars) and purple sea urchin (white bars) by year (error bar indicates SE), and B, D, and F) relative size distribution of red sea urchin, from survey data.

gon are not temporally stable, and evidence of consistent annual fishery recruitment was not found in the years covered by this work.

Fishery Trends

In their 1997 analysis of California's red sea urchin fishery, Kalvass and Hendrix (1997) observed a dynamic response pattern, where the fished stock responds to fishing effort and vice versa, per Shepherd (1993) (Fig. 21 in Kalvass and Hendrix, 1997) and they were curious if this downward trend would continue.

Oregon's red sea urchin fishery exhibited a nearly identical "dynamic response" pattern through the early years of the fishery. Further, the longer time-series presented here, shows this pattern moved to stabilization following a level threshold fishery effort for many years (Fig. 18). Similar to the Kalvass and Hendrix (1997) analysis, the data presented here show a fluctuating pattern in the relationship between stock and fishing effort; however, these data show that a somewhat stable pattern has emerged (see years 1996 to 2022: Fig. 18). Recent increases of red sea urchin densities appear likely to make a near-term shift in the pattern of this relationship.

Value of Market Sampling

When evaluating size structure of red sea urchin stocks, managers employ two methods: 1) market sampling and 2) population surveys. While population abundance surveys are costly, the cost is acute and the data provides a fishery independent assessment of stocks. Market sampling is easily executed (e.g., can be performed dockside by a single individual); however, the resulting data is fishery dependent. The primary difference between these two datasets is that the fishery independent data is unbiased by fishery selectivity about the stock.

Comparing synchronized market sampling and population survey size structures shows that modes of legal sized red sea urchins are congruent; however (expectedly), only the population survey data gives an indication

of pre-fishery recruitment, a key component of evaluating stock status. Understanding pre-fishery recruitment is especially important to understanding contemporary stock conditions in cases (such as red sea urchin) where it may take many years prior to settlement to reach MLS. While market sampling may provide a good indication of size distribution of the stock in years when recruitment hasn't occurred recently (1993, 2011, 2014), when there are recent recruitment events that have not recruited into the fishery, size structure from the two methods is drastically different (2016, 2019) (Fig. 19).

Overall, market sampling data has provided some value. Detection rates of large, old sea urchins in market samples can provide insight to the size structure of the overall population. In addition, working with and communicating with industry is highly valued. Consideration should be given to the comparative costs of surveys vs. market sampling.

Table 4.—Key management actions for sea urchin in Oregon, thru 2022.

Year	Management action
Prior to 1988	No specific permit required.
1988	Permit system adopted by Oregon Fish and Wildlife Commission (OFWC). Number of permits set at 92, issued to individual divers, 9.07 t (20,000 lb) landing requirement over previous two years.
	Adopted minimum size (76.2 mm (3 in) test diameter), minimum harvest depth (3.3 m (10 ft MLLW)), maximum number of divers in water per boat (2), and a logbook requirement.
1989	OFWC reduced the number of permits to 46, through attrition (failure to renew). Changed the 9.07 t (20,000 lb) landing requirement from a 2 yr to a 1 yr requirement.
	Restricted the maximum number of non-permitted people on a urchin boat to two.
	Medical transfer rule, allowing permitees to temporarily transfer their permit if an injury or illness was suffered, made permanent.
1990	OFWC established buffer zones around three key Steller sea lion pupping rocks (Seal Rock and Long Brown Rock on Orford Reef and Pyramid Rock on Rogue Reef).
	Amended medical transfers to:
	The greater of either the lb taken the previous year or 9.07 t (20,000 lb).
	A limit on each medical transfer to 90 days
	No limit on the number of transfers
1991	OFWC raised the minimum size to 88.9 mm (3.5 in) test diameter and reduced the allowable number of undersized urchins to 50 per landing.
	OFWC adopted a 50.8 mm (2 in) minimum size limit on purple sea urchins, and a special harvest permit, requiring pre-harvest surveys
1992	Due to Steller sea lion interactions coupled with poor market quality of sea urchins during the summer OFWC established a sea urchin season at Orford Reef (May–October).
1993	OFWC established subtidal reserves at Gregory Pt. and Pirates Cove.
1994	OFWC closed urchin harvest on Orford Reef from May 1 to October 31.
1995	New permit system adopted by OFWC including:
	New target level number of permits (30)
	New annual renewal poundage (2.27 t (5,000 lb))
	Reinstated 2 yr continuous medical transfer limit
2014	Permit lottery suspended for two years while fishery is reviewed.
2016	Permits reduced to 12, mixed gas diving disallowed, sea cucumbers included to permit

Fishery Management

Worldwide, fisheries for sea urchins have a poor record of sustainability and have generally followed a trend of quick expansion followed by an equally rapid decline (Andrew et al., 2002). Oregon's red sea urchin fishery is well characterized by this assessment for most of its history. The recent massive episodic recruitment of red sea urchin occurred while populations were at historically low levels, reducing Oregon's future management prioritization of maintaining threshold densities at fishing areas.

The three key methods of Oregon's red sea urchin fishery management (effort limitation, reserve areas, and MLS) have variable levels of efficacy. Effort limitation has worked well to assure investment and encourages consistency of fishery catch. Reserve areas (direct and de facto) enhance accurate stock assessment and set aside some areas to promote conditions of reproduction (by providing high density stocks absent of fishery pressure). Minimum size limits in this fishery create a size reserve; however, I suspect that the efficacy is limited (as a method of assuring stock sustainability) given the long lives and episodic nature of recruitment.

The Future of Oregon's Sea Urchin Fisheries

Unlike conventional "recruitment fisheries" where a surplus of new fishery recruits are somewhat reliably available on an annual basis, the red sea urchin stock and fishery appears to depend on a longer time scale. Oregon's red sea urchin fishery has been fueled by only a few recruitment events over its 36-yr history. This key temporal difference from conventional fisheries must be considered in management. Management actions such as use of reserves, effort limitation (despite stock surpluses), and periodic fishery independent population monitoring (using index sites rather than fully randomized designs) appear more appropriate than annual quotas for Oregon's red sea urchin fishery.

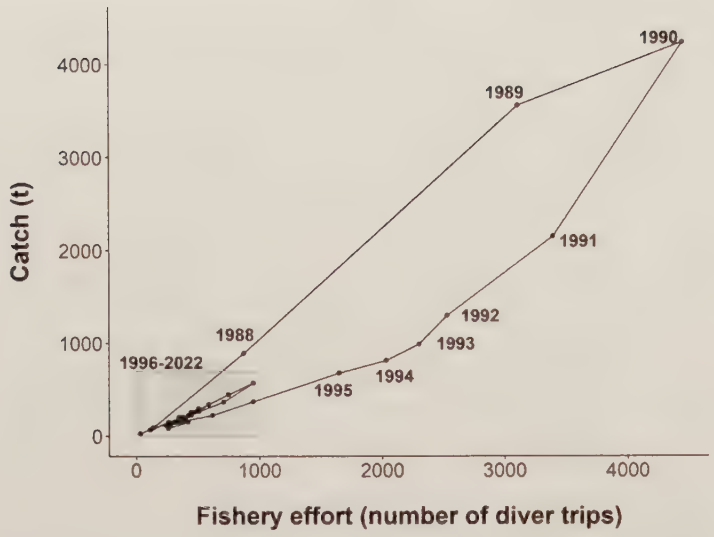


Figure 18.—Red sea urchin, catch (t) compared with effort (vessel trips) in Oregon, 1986–2019, exhibiting Shepherd's (1993) dynamic response pattern. Note rapid acceleration of both catch and effort in early years of fishery (1986–90), rapid decline for both (1991–96), then stability at relatively low levels (1996–2022). This pattern is characteristic of boom-bust fisheries.

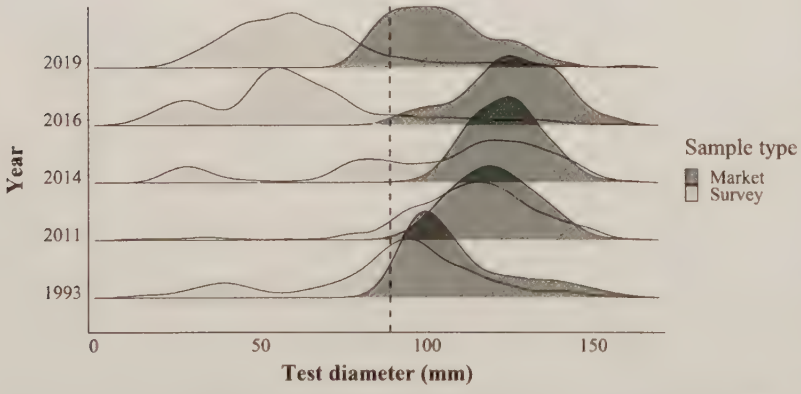


Figure 19.—Size distribution of red sea urchin, from fishery market samples (dark shaded) and population surveys (light shaded) at Orford Reef.

The future of Oregon's red sea urchin fishery is unclear. As of 2019, stocks were at all-time highs while kelp abundances were at low points (Hamilton et al., 2020). It seems likely that the recent episodic recruitment events may fuel the fishery for many years to come, given that the previous event sustained the fishery for more than 30 years.

Ecologically, there are serious concerns of the population boom of both red and purple sea urchins along the U.S. west coast. The boom is particularly acute, given the absence of sea urchins' primary predator at both adult (sea otters, *Enhydra lutris*) (Jameson et al., 1982) and juvenile life stages (sunflower sea stars, *Pycnopodia helianthoides*) (Harvell et al., 2019). Particularly, there are concerns of how the robust sea urchin population may depress kelp beds and possible conversion from a "healthy kelp bed" state to an "urchin barren" state. The restoration of sea otter and sunflower sea star populations is currently being considered and may serve as an effective and natural method of stabilizing nearshore ecosystems (Rogers-Bennett and Catton, 2019).

Lastly, purple sea urchin populations have boomed on the U.S. west coast and managing these massive populations is challenging. Populations at Orford Reef survey sites (which were not designed to assess purple sea urchin since they are typically in shallower zones) have gone from barely detectable (<0.01–0.10/m^2 1984–2014) to very high (6.24/m^2 in 2019), to estimates in the neighborhood of 350 million individual purple sea urchin (Groth[1]). While this surplus may be inviting to consider new fishery options, history has shown low market appeal, despite high abundance. The biological need of reducing these populations is unclear, but most stakeholders encourage active management, such as direct removal or destruction. Given their long life history and persistence, it seems unlikely this will resolve quickly by natural means.

Acknowledgments

Data and accounts in this manuscript consists of many generations of biologists and sea urchin divers' careers, and this author was only a small part.

To this point, I thank prior generations for their hard work surveying and documenting the many aspects involved including: N. Richmond, J. Golden, J. McCrae, J. Schafer, C. Wood, N. Wilsman, A. Merems, J. Heinrich, and B. Miller. I thank my collaborator and friend Steven Rumrill for his help in working for and guiding this research. Kendall Smith, Dave Fox, and Bob Hannah helped shape this manuscript. This journal's editors and two anonymous reviewers provided helpful comments.

[1]Groth, S. D. 2019. Sea urchins, sea stars, abalone and kelp beds. Unpubl. rep., Oreg. Dep. Fish. Wildl., Charleston, OR, 2 p.

Thanks to generations of sea urchin divers who made this work possible in a number of ways. Foremost, the crew of the F/V *Mach 1* including Tim Foley, Mike Ashmon, Tom Butterbaugh, James Weinrich, Mark Gmeiner, Nick Foley, and others were instrumental in performing surveys, collecting data, and shaping their fishery. Others industry members such as H. Allen, K. Hiersche, Vicki and Scott at NorCal, and several others facilitated fishery understanding and sampling. Sea urchin diving is a beautiful form of fishing full of skill, adventure, hard work, and risk. Finally, I dedicate this manuscript to ODFW Marine Biologist Neil Richmond. I only met Neil once; however, I appreciate his careful care for this fishery in its early years, managing the challenges of a new, booming fishery.

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Abalone (Haliotidae) in Oregon: Trends in Populations and Fisheries

SCOTT GROTH and KENDALL GROTH

Introduction

Biology

Abalones, marine mollusks from the family Haliotidae, are generally found in sparse populations constrained by their narrow habitat requirements and life history parameters. They feed on marine vegetation found in rocky nearshore areas. Globally, these rocky vegetated habitats, which are ideal for abalone populations, are uncommon; strongholds for populations include the nearshore kelp beds of western North America, South Africa, Japan, Australia, and New Zealand (Lindberg, 1992). Abalones are dioecious and reproduce by broadcast spawning, where males and females simultaneously emit gametes into the water column. When fertilization is successful, resulting larvae have a short free-swimming phase prior to benthic settlement, suggesting limited dispersal (Miyake et al., 2017). Reproductive success may be limited when individuals are not in close proximity; higher density stocks are found to have higher fertilization rates (Babcock and Keesing, 1999). Given the low probability of reproductive success, abalones have adapted long life spans (Tegner et al., 1996; Rogers-Bennett et al., 2007; Leaf et al., 2008). Low density populations of abalones have been shown to depend on spatial aggregation, sheltered embayments, and consistent kelp availability (Hart et al., 2020). The paucity of suitable habitats with favorable environmental conditions combined with their life history constrain abalone spatial distribution and abundance.

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Fishery History

Abalones are a highly desirable seafood and have been used by many cultures. Native American tribes throughout the U.S. west coast treasured abalones not only as a food source but also as one of the most prized shells in trade (Heizer, 1940; Bonnot, 1948). Beginning in the 1850's, Chinese immigrants developed intertidal commercial fisheries for California abalones, where catch rose to over 1.8 million kg in 1879. That fishery gave way to a dive fishery undertaken by Japanese immigrants. In turn, the Japanese fishery gave way to the Caucasian immigrant fishery beginning in 1929. By the 1940's large scale commercial fisheries of California abalones were principally undertaken by Caucasians (Cox, 1962). While this describes activities central to abalone population, little documentation is available for early fisheries in Oregon.

Commercial fisheries occurred for all abalone species along the U.S. west coast, each showing a "boom and bust" pattern. By 1969, commercial fishing for abalones in California slowed down, then closed in 1997 (CDFW, 2005). A small commercial fishery for flat abalone, *Haliotis walal-*

ABSTRACT—Abalones (Haliotidae) are highly prized and eagerly sought; however, their populations are particularly sensitive to fishing and environmental changes. All seven abalone species living along the U.S. west coast are imperiled to some degree. Historically, four species of abalones have occurred in Oregon, and three are currently found. Along the U.S. west coast, fisheries have occurred for some of these species; however, all were suspended as of 2018. Red abalone, Haliotis rufescens, *are at the northern extent of their range in Oregon, and populations are sparse but individual size is very large. Red abalone were fished commercially (1958–62) and recreationally (1900's–2018), but both were closed due to sustainability concerns. Oregon's recreational red abalone fishery began as a "food" fishery, but recognizing their large individual size, lowering catch limits, and increasing scarcity eventually moved to a "trophy" fishery. Oregon's red abalone are sparse, but population levels have varied widely; environmental changes including tsunami and coastal flooding in 1964 and oceanographic changes in the late 2010's have severely affected their abundance.*

Southern Oregon (nearshore waters from Coos Bay to California/Oregon border) is a stronghold for populations of flat abalone, H. walallensis, *a smaller species which is naturally rare throughout their range. They were the subject of a small but valuable artisanal commercial fishery (2001–08), which closed due to sustainability concerns. Oregon's commercial flat abalone fishery showed clear effects of fishing, despite its small scope and short duration. Since the closure of the fishery, stocks appear to have further declined, probably owing to environmental changes and latent fishing effects.*

Pinto abalone, H. kamtschatkana, *and black abalone,* H. cracherodii, *have also been found in Oregon, but in very low numbers and not consistently over time. Overall, abalones in Oregon have undergone large shifts across the past century. Oregon's abalones are currently imperiled and there is interest to both maintain their populations and allow fishing opportunities. Finding ways to utilize societal investment in Oregon's abalones may be key to assuring sustainability.*

lensis, was open in Oregon from 2001 to 2008, detailed later. Recreational fishing for red abalone, *Haliotis rufescens*, endured in Oregon but ended in 2017 (Groth[1]).

Abalone populations are particularly sensitive to fishing, confounding sustainability of their fisheries. Abalones are often sessile and occupy shallow depths, limiting refugia from fishing. Typically, fishing effort focuses first on areas of high densities, which, in the case of abalone, are spawning aggregations containing the most critical individuals to the population. Once spawning aggregations are fished down, serial depletion is likely (Karpov et al., 2000; Hobday et al., 2001; Hilborn et al., 2005).

Consequences of serial depletion are evident in the many abalone fisheries which have collapsed. Along the U.S. west coast, two of the seven abalone species (*H. cracherodii* and *H. sorenseni*) are listed as "endangered" under the Endangered Species Act (ESA).[2] The remaining five west coast abalone species are listed as "critically endangered" or "endangered" with the International Union for Conservation of Nature (IUCN[3]).

Abalone in Oregon

In Oregon, four species of abalone have been found within the state's biogeographic range: red abalone, *Haliotis rufescens*; flat abalone, *H. walallensis*; pinto abalone, *H. kamtschatkana*; and black abalone, *H. cracherodii* (Fig. 1). Abalones are principally found in southern Oregon where rocky nearshore kelp beds are prevalent but discontinuous. Red abalone are found as far north as Coos Bay, Oreg., though a few individuals have been found as far north as Depoe Bay (McCauley and Marriage, 1955). Flat abalone are found throughout southern Oregon and northward in areas around Depoe Bay.

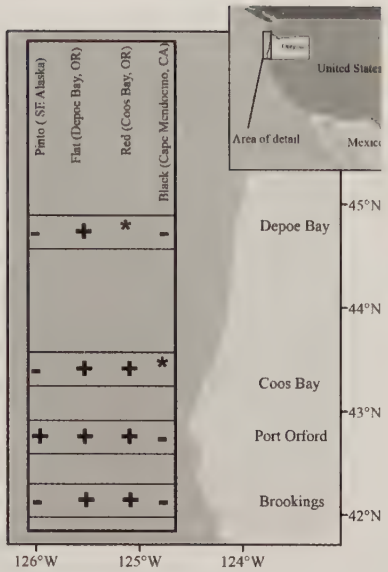
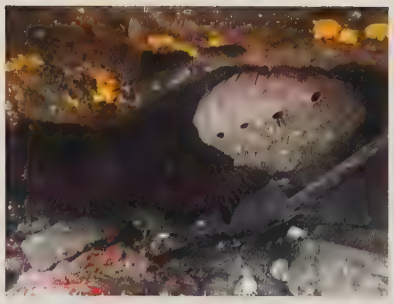


Figure 1.—Condition of known presence of four abalone species (black, red, flat, and pinto) in Oregon at key locations. Plus signs indicate definite presence, minus signs indicate no records of finding, and an asterisk indicates special conditions described in sections for each species.

The geographic range of the flat abalone is not well documented; some publications list Newport, Oreg. (Rogers-Bennett, 2007), while others extend their range well into Washington (Geiger, 1999). Because morphological identification is most common, the phenotypic plasticity found for flat and pinto abalone (Straus, 2010) may confound published range extents, especially for flat abalone. Pinto abalone range from southeast Alaska to Point Conception, Calif. (Geiger, 1999; NMFS, 2014); however, they are extremely rare in Oregon. The northern range termination of black abalone, *H. cracherodii*, is currently Point Arena, Calif. (NMFS, 2018); however, the northern termination of the historic range may extend to Coos Bay. (Marriage[4]).



Red abalone (right) and red sea urchins (left) found under a rock near Port Orford, Oregon. Photo: Scott Groth.

Red Abalone

Red abalone occur cryptically in nearshore kelp beds of southern Oregon. Their populations have been affected by fisheries, ecological changes, stock enhancements, and devastating environmental events. Red abalone have been reported in the Brookings, Oreg., area since at least the early 1900's (McCauley[5]). Since that time, their northern range termination was extended incrementally until it was defined as Coos Bay, Oreg., in 1957 (Lyons, 1957). Red abalone are the largest species of abalone in the world (Cox, 1962) and are especially prized in fisheries. The largest specimens of red abalone have been found in Oregon. A brief exploratory commercial fishery existed from 1958 to 1962, while a recreational fishery persisted from 1953 to 2017.

Ecological changes which have affected populations of red abalone include the removal of sea otters, *Enhydra lutris*, a key abalone predator in the late 1800's (Jameson et al., 1982), and in recent years increased spatial competition by sea urchins (Groth, 2023) and lowered food abundance (Rogers-Bennett and Catton, 2019). Given their rarity and desirability for expanded fisheries, two major stock enhancement efforts have been made. Lastly, freshwater influx and ocean warming events in recent history have

[1]Groth, S. March 16, 2018. Modifications to the red abalone fishery. Oregon Fish and Wildlife Commission, Salem, OR (avail. at https://www.dfw.state.or.us/agency/commission/minutes/18/03_Mar/index.asp).

[2]https://www.fisheries.noaa.gov/species-diectory/threatened-endangered

[3]IUCN (International Union for Conservation of Nature). 2023. Available at https://www.iucnredlist.org/, accessed 23 Apr. 2023.

[4]Marriage, L. D. 1954. Distribution of abalone in Oregon. Oreg. Fish. Comm. Unpubl. internal memo.

[5]McCauley, J. E. 1953. The mussel, piddock, and abalone resources of Oregon's outer coast. Oreg. Fish. Comm. unpubl. internal memo.

likely had substantial impacts on Oregon's red abalone population.

Fisheries

Exploratory Commercial, 1958–62

From 1958 to 1962, the Oregon Fish Commission (OFC) worked with contract divers to explore the commercial fishing of red abalone in exchange for data collection. Data collected from this effort included: 1) dive reports describing sites, abundance, habitat, spatial competitors, etc. and 2) length measurements of abalones. From 1958 to 1959, at least nine abalone were taken during 27 dives. In 1960, at least 102 red abalone were taken in 31 dives. During 1961–62, few dives were performed as momentum for this project waned. Over the term of the contract, at least 111 red abalone were taken, then sold or shared. After four years, this work yielded knowledge of red abalone distribution and biology. In 1962, the Oregon Fish Commission (OFC) staff recommended cancellation of this commercial fishery until greater concentration of red abalone were found (Snow[6]). To date, no further commercial fishing for red abalone has occurred in Oregon.

Recreational Fishery and Management, 1950's–2021

Oregon's recreational red abalone fishery has grown in popularity over time, despite fishery rules becoming more strict due to increased participation and concern for stock depletion. Recreational red abalone fishing began in Brookings prior to the early 1950's (McCauley et al., 1955). In 1959, Oregon adopted its first rules for the fishery: 1) 203 mm (8 inches) minimum legal size (MLS), 2) required to measure abalone before removing, and 3) limit of 3 abalone per every 7 days (Snow[7]). In 1973, a red abalone catch reporting permit was introduced to elucidate catch rates (OFC[8]); the permit remained in place until 1979. In 1996, rules were established that 1) reduced the daily limit to 1 abalone, 2) reduced the annual limit to 5, and 3) added the requirement of a catch reporting permit. In 2018, the Oregon Fish and Wildlife Commission (OFWC) suspended the fishery for a period of 3 years (Groth[1]). In 2021, the recreational fishery was suspended an additional 3 years (Rumrill[9]). In each case, management actions responded to concerns about red abalone population levels, viewed by fishery participants and managers of each era as further imperiled.

Fishermen with abalones from Oregon's commercial red abalone fishery in the vicinity of Brookings, Oregon. Photo: ODFW Archives.

Lee Narin (commercial fisherman) and Dale Snow (OFC biologist) posing with a red abalone during Oregon's explorative commercial red abalone fishery. Photo: ODFW Archives.

Catch Report Permits

Catch report permits were required (at no cost to participants and not limited in issuance) in the recreational red abalone fishery during 1973–79 and 1996–2017. Permit data included information about catch (i.e., number taken, location, depth, method, etc.) and demographic information about the permittee. Issuance of each year's catch permit required submission of the prior year's catch data.

In 2013, the catch report permit was expanded to also include rock scallops, *Crassodoma gigantea*, often collected by the same people who target red abalone. A census of 2014 permittees found 15% targeted rock scallops only (Gregory[10]). We used this estimation to isolate permits used to target red abalone for years 2013–17, when both species were on the same permit.

From 1973 to 1979, on average, 126 catch reporting permits were issued, compared to 175 from 1996 to 2017. Permit issuance trended up sharply during 1996–2017, peaking in 2017 at 280 permits (Fig. 2).

Catch Data

To develop catch rates, reported permits were tabulated, then expanded to account for non-reported permits. A total of 4,946 permits were issued and 2,692 were submitted. Non-reported catch was accounted for by expansion using rates of reporting and catch by year using methods from a sepa-

[6]Snow, D. 1962. Abalone research studies 1958–1962. OFC unpubl. memo.

[7]Snow, D. 1959. Suggested personal use regulations for red abalone (*Haliotis rufescens*) in Oregon. OFC unpubl. memo.

[8]Oregon Fish Commission. 1972. Staff request for a permit system to harvest red abalone. ODFW unpubl. memo.

[9]Rumrill, S. February 12, 2021 Suspension of recreational abalone fishery. Oregon Fish and Wildlife Commission, Salem, OR (avail. at https://www.dfw.state.or.us/agency/commission/minutes/21/02_Feb/index.asp).

[10]Gregory, K. 2018. Estimates of total harvest of red abalone in Oregon. ODFW unpubl. memo.

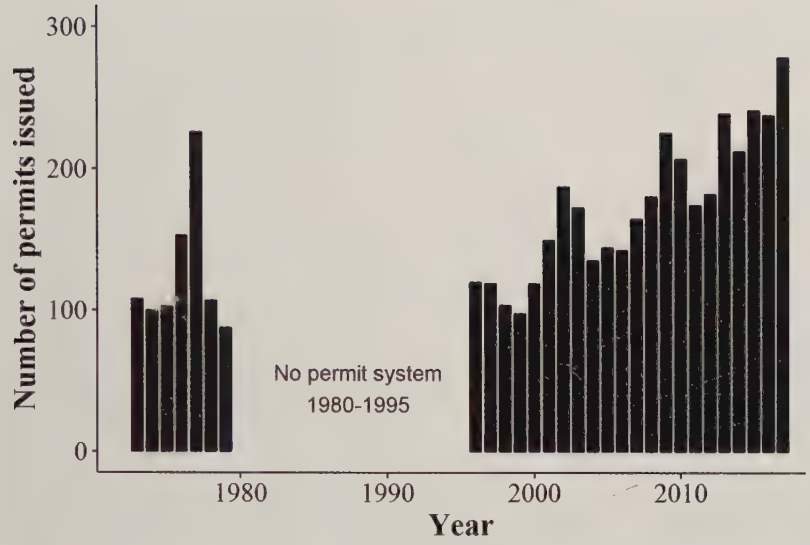


Figure 2.—Oregon recreational red abalone permits issued from 1973 to 1979 and 1996 to 2017.

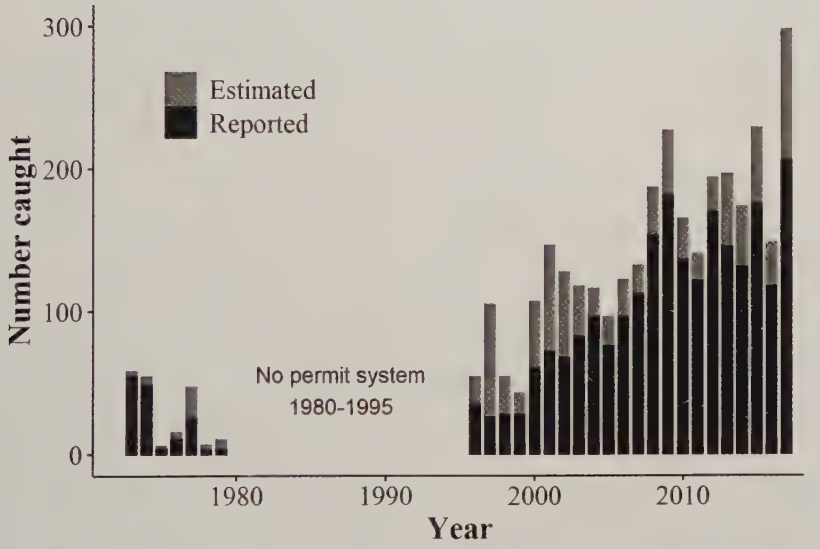


Figure 3.—Oregon recreational red abalone fishery catch from 1973 to 2017.

rate study (Gregory[10]). To quantify the total catch, reported and non-reported catch were combined. Total annual harvests have increased over time owing to higher effort and success rates. From 1973 to 1979, an average of 28 red abalone were taken, then from 1996 to 2017 the average increased to 145, peaking in 2017 when 299 red abalone were taken (Fig. 3).

Harvest Methods, Depth, and Area

Harvest methods for red abalone in Oregon include: 1) scuba, 2) free diving (snorkeling or breath-hold diving), and 3) shore picking (i.e., tide pooling or rock picking). Total catch among all years (1973–79 and 1996–2017) was 2,494 red abalone. Of these, 51.4% were taken using scuba, 29.3% by free diving, and 19.2% from shore picking (n=2,494). Mean depth of catch was 5.2 m for scuba, 4.0 m for free diving and 0.3 m for shore picking (n=2,169).

Beginning in 2005, permittees were required to report harvest areas and activity was strongly focused on southern Oregon. Most catch (95.4%) was reported (n=2,221) near Brookings, while 3.3% was reported in Port Orford and less than 1% was reported at areas near Gold Beach and Coos Bay (Fig. 4).

Catch Size

In Oregon's recreational red abalone fishery, participants are primarily motivated to take trophy-sized shells, not simply for harvest of meat (personal observ.). Conventionally, a trophy-sized red abalone shell is greater than 254 mm (10 in), and many participants go so far as to carry measuring devices (typically fixed arm calipers) capable of assessing this size rather than simply meeting the minimum legal size (MLS).

Owing to the motivation for trophy shells, the fisheries' MLS did little to govern catches. The MLS of red abalone is 203 mm shell length (SL), but the fishery mean size was 245 mm SL. While this may seem like a minor difference of the linear shell measurement, by weight, the two examples would be 1.6 and 2.8 kg. The large mean size indicates larger animals were targeted and available; however, the lack of new recruitment and availability of smaller animals may also contribute.

By method, scuba and free divers found the largest red abalone. We examined mean shell lengths from two eras using a Mann-Whitney U (MW) test, since each era's shell length data is independent and non-parametric. The mean shell length of red abalone (all methods) significantly increased from the 1970–79 era (230.5 mm) to

Table 1.—Mean shell length (and standard deviation) of red abalone in Oregon from catch reporting permits by era 1970–79 and 1996–2017.

	Scuba		Shore Pick		Free dive		Total	
Era	Mean size	Count	Mean size	Count	Mean size	Count	Mean size	Count
1970–1979	233.2 (19.8)	113	221.5 (9.2)	9	221.7 (19.6)	26	230.5 (19.8)	148
1996–2017	247.2 (17.6)	1,137	232.8 (19.6)	429	251.4 (17.9)	703	245.8 (19.2)	2,269
Avg.	246.0 (18.3)	1,250	232.6 (19.5)	438	250.3 (18.7)	729	244.9 (19.6)	2,417

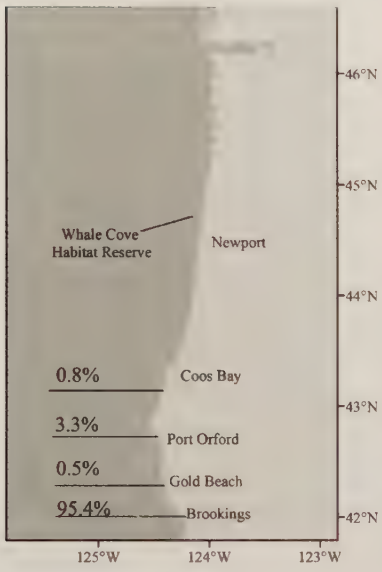


Figure 4.—Percentage of reported recreational red abalone fishery catch by area from 1996 to 2017 in Oregon (n=1,894). Note: only one red abalone was reported as caught north of Coos Bay, Oregon.

the 1996–2017 period (245.8 mm) ($p<0.0001$ MW test) (Table 1).

Surveys

In the 1950's, soon after the documented geographic range of red abalone was extended into Oregon, investigations began to develop further understanding. From 1958 to 1962, the OFC contracted with divers to survey red abalone. The nature of the surveys was qualitative, though communications and measurements of catch improved understanding of the population. From the 1960's to 1990's, the OFC (later renamed the Oregon Department of Fish and Wildlife (ODFW)) performed qualitative surveys and worked with the public to continue building an understanding of Oregon's red abalone. Quantitative surveys for red abalone were first attempted in 2011, formalized in 2015, then repeated in 2019 and 2022.

Pilot Surveys, 2011

In 2011, a 2-day experiment tested two methods to quantify density of red abalone and describe their size distribution. These methods were 1) polygonal index sites and 2) timed surveys. These methods were tested in an area near Brookings, the most important site to the recreational red abalone fishery.

On 23 June 2011, to test a method for red abalone density measurement, a 77 hectare polygonal index site was developed from the shoreline to a 10 m bathymetric line, encapsulating the most heavily used harvest area. Random points were drawn to begin subtidal belt transect lines and ten, 30 x 2 m transects were performed. The result of this survey was that zero red abalone were found within transects. A key takeaway from this result was that even in the most dense locations in Oregon, red abalone may be too rare to practically perform simple random sampling methods.

On 24 June 2011, to focus on size distribution of red abalone, divers performed timed surveys. While timed surveys have not proven to be an accurate density metric (Gorfine et al., 1996), they can provide robust size distribution data at low density locations. A team of two divers was deployed at sites where the fishery operates (red abalone were expected to be present). Each diver recorded the number and size (SL) of abalones found. The result of this survey was 55 shell length measurements and a count-per-hour density measurement.

Index Surveys in 2015, 2019, and 2022

In 2015, index sites were selected by identifying areas where the fishery operated (near Port Orford and Brookings) and therefore a relatively high density of abalones was expected. Teams of two divers performed 30 x 2 m subtidal belt transects, identifying and enumerating all abalones and sea urchins. Forty-four transects were performed during 5 days, 20 in Port Orford (Fig. 5a) and 24 in Brookings (Fig. 5b). The index survey method provided a measurement of both density and size distribution of abalones at each site. In 2019, 11 sites in Port Orford were repeated and in 2022, 13 sites in Brookings were repeated. This method is biased towards high density locations since transects are performed where the fishery operates. In this way, it may only be appropriate to compare relative abundances found at surveys temporally.

Trends in Abundance

Abalone abundance is not easily measured, and methodology in Oregon have been sparsely assessed and inconsistent over time, confounding comparisons. Surveys focusing on capturing relative abundance, a measure of density used to index populations, to determine the total number of abalones in Oregon (absolute abundance) has been seen as too costly. Early surveys were simply timed dives with recorded notes about how many abalones were found in each dive.

Timed survey data from 1969 and 2011 show a much higher detection rate in recent years; however, these results could easily be affected by diver expertise, a key problem with timed surveys (Table 2). The results of 2015 index surveys showed

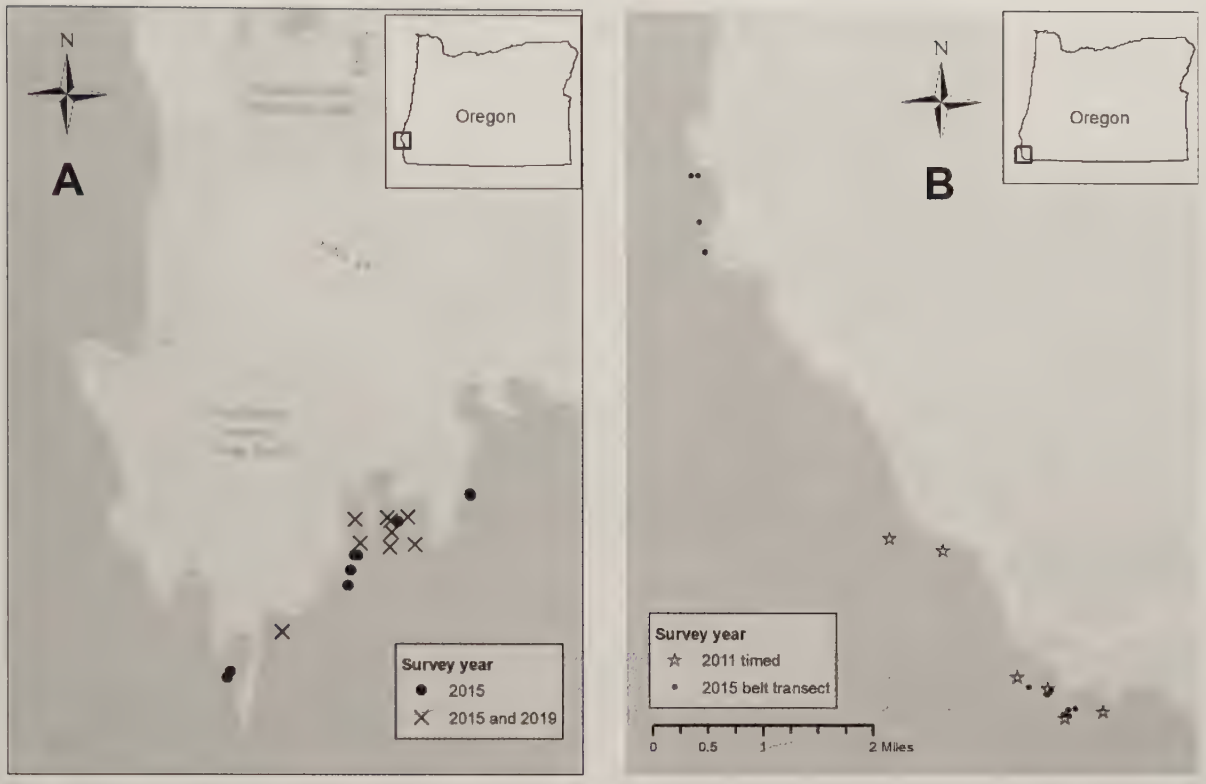


Figure 5.—Abalone survey sites at A) Port Orford, Oreg. (belt transects) and B) Brookings, Oreg. (timed and belt transects) for 2015 and 2019.

Table 2.—Number of red abalone seen in timed dives by year (1969, 2011) in Brookings, Oreg.

Year	No. seen	Survey min	Abalone/hr
1969	46	250	11.15
2011	55	127	24.39

Table 3.—Red abalone densities (per m^2) from fishery independent index sites in 2015, 2019, and 2022 by port in Oregon.

	Port	
Year	Brookings	Port Orford
2015	0.047	0.030
2019		0.017
2022	0.014	

low density of red abalone at both Port Orford and Brookings (0.047/m^2 and 0.03/m^2, respectively), subsequent surveys (Brookings in 2022 and Port Orford in 2019) showed further lowered densities at both sites (0.014/m^2 and 0.017/m^2, respectively) (Table 3).

Size Distribution Trends

All qualitative and quantitative surveys measured only emergent abalones (i.e., those found without use of lights or by moving substrate) optimizing comparison of size distributions despite methodological differences. Pooled (state-wide) red abalone size distributions have been relatively similar throughout time; however, some changes became evident in 2019 (Fig. 6).

Mean SL of red abalone among area by year has not changed significantly ($p>0.1$ in all comparisons in Brookings (MW test)), which include data up to 2015, (Table 4), however Port Orford size length data has significantly changed (reduced) in recent years, which include 2019 surveys ($p=0.02$ MW test). It seems likely that the change in size distribution found in 2019 is due to detection rate, where smaller abalones have come out of cryptic habitats in search of kelp, given low kelp abundances in recent years.

Enhancement Projects

Given the difficulty of maintaining sustainability in abalone fisheries compared to the ease in which abalones are cultured in a laboratory, stock enhancement is an inviting opportunity. Worldwide, the most large-scale and successful abalone stock enhancement efforts have occurred in Japan, with 10 million seed cultured and distributed in one year (1979) alone (Leighton 1989; Tegner and Butler, 1989). Although abalone seed have been successfully hatched and distributed, juvenile survival rates vary based on species and geographic circumstances. Tegner and Butler (1989) attempted to apply Japanese seeding techniques to southern California red,

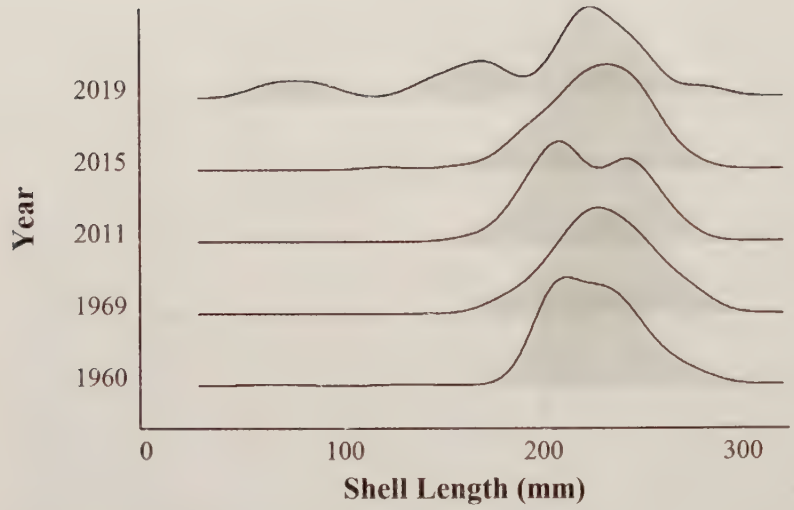


Figure 6.—Red abalone shell lengths from Brookings, Oreg., survey data 1960, 1969, 2011, 2015, and 2019.

Table 4.—Mean shell length (and standard deviation) of red abalone in Oregon by port from survey data for 1960, 1969, 2011, 2015 and 2019.

	Area			
Year	Brookings	*n*	Port Orford	*n*
1960	225.5 (25.5)	221		
1969	230.1 (23.9)	86		
2011	221.9 (24.5)	55		
2015	225.7 (23.8)	51	222.6 (30.2)	33
2019			196.9 (56.1)	26

green, and pink abalone populations but had minimal success, and another enhancement experiment in northern California also found low juvenile red abalone success rates (Rogers-Bennett and Pearse, 1998). ODFW has made two efforts to bolster red abalone stocks, a 1960's project at Whale Cove and a 1990's effort in southern Oregon (Golden[11]).

Whale Cove, 1967–2018

ODFW attempted to create new spawning stocks of red abalone to develop a new fishing opportunity near Whale Cove, a small area just south of Depoe Bay and 100 miles north of their northern range extent. Hatchery-reared juveniles and transplanted adult red abalone were introduced into the small cove due to its protected rocky habitat, abundance of marine vegetation, and ease of access. Conceptually, a protected Whale Cove red abalone population would serve as a spawning stock for adjacent catch areas and provide experimental data on growth and reproduction. Accordingly, Whale Cove was designated as a Habitat Refuge (HR) in 1967, disallowing take of shellfish (Nielsen[12]). Staff introduced red abalone from 1967 to 1974, then surveyed the resulting population from 1967 to 2018.

From 1967 to 1974, 5,660 juvenile and 277 adult red abalone were introduced to the intertidal areas of Whale Cove HR. First, in May 1967, 660 hatchery-reared juvenile red abalone were introduced; those abalone survived, then in July of 1967 an additional 5,000 juveniles (5–20 mm SL) were planted. Then, in 1968–69, 261 adult red abalone collected at Fort Bragg, Calif., were placed in Whale Cove. The last outplanting occurred in March 1974 (16 adult red abalone). To track populations, red abalone were tagged. In total, 576 red abalone were affixed with stainless steel tags secured using wire through their respiratory holes. Intertidal surveys of red abalone within Whale Cove were performed annually (1968–91) then sporadically (1992–2018) to measure mortality and growth.

The Whale Cove red abalone project provided some baseline information about growth, age, longevity, and reproduction rates in Oregon. It was reported that growth rates of these red abalone to legal harvestable size (203 mm SL) could be as little as 10 years (Lukas[13]), while earlier calculations from the same study suggested that legal size red abalone would be around 22 years old (Gaumer[14]). Red abalone from this initial planting were found up until 1991 when annual surveys were suspended. These red abalone were most likely from the 1967 planting, then being 24 years old (n=12, mean size 190 mm SL). Further, surveys in 2009 (n=4, mean size 208 mm SL) and 2017 (n=4) found 4 adult red abalone (Metzler[15]) which were almost certainly from the original planting (at the same locations as recalled by historic staff) that are likely as old as 50 years. Recapture rates estimated that 241 of the 5,500 (4.3%) red abalone planted in Whale Cove persisted in 1975, eight years after planting (Gaumer[14]). Throughout this study, no evidence of new, wild reproduction was found (no new recruits were found in periodic searches). A few tags (apart from shells) and shells with tags were found in the modern era (2005–17) demonstrating recent demise; however, no live animals with tags have been found recently (personal observ.).

[11]Golden, J. T., and C. Langdon. 1995. Development of a red abalone broodstock from abalone native to Oregon, abalone culture and outplanting experiment. ODFW unpubl. memo.

[12]Nielsen, J. 1967. Whale Cove shellfish regulations for Fish Commission public hearing. Oreg. Fish Comm. Hearing.

[13]Lukas, J. 1978. Abalone research and management activities, 1958–77. ODFW unpubl. memo.

[14]Gaumer, T. F. 1975. Whale Cove abalone. ODFW unpubl. memo.

[15]Metzler, J. 2017. ODFW abalone snorkel surveys. ODFW unpubl. memo.

OFC biologist tagging red abalone at Whale Cove, Oregon. Photo: ODFW Archives.

Flat abalone, surrounded by encrusting invertebrates in Nellies Cove, Oregon. Photo: Scott Groth.

Ultimately, this project had no noticeable effect in creation of a new fishing opportunity for red abalone. From catch permit data (1972–79 and 1996–2017), only one red abalone was taken north of Coos Bay among the 2,221 reported.

Southern Oregon Enhancement Project, 1994–99

In 1994, another red abalone outplanting project was developed, a cooperative effort between ODFW and Oregon State University (OSU) (Golden[16]). The goal of this project was to develop rearing methods for red abalone that were locally adapted, making new populations to serve as a local spawner source. The project was delayed due to infection concerns (i.e., renal coccidian, withering foot) due to importation of hatchery stock from California. The program then shifted objectives to creating a broodstock population from red abalone collected in Oregon. Approximately 20 red abalone were collected by staff and contracted divers. Spawning was successful and approximately 50 juvenile red abalone were reared until they reached outplanting size. The project was halted in 1999 due to lack of funding and momentum (Langdon[17]). In 2002, many red abalone remaining from this program were outplanted near Charleston by commercial sea urchin divers (Foley[18]), though no follow up surveys occurred.

Environmental Events

In the nearly 70-year historical record of reports regarding red abalone in Oregon, two environmental events were coincidental to major reductions in red abalone populations. First, in 1964, a tsunami and coastal flooding are likely to have reduced abalone populations substantially. On March 27, 1964 a high-magnitude earthquake, originating in Alaska produced severe waves and current in southern Oregon (Witter et al., 2001), likely strong enough to dislodge or destroy intertidal fauna. During flooding, massive sedimentation from flooding was exacerbated by the adjacent construction of Highway101. ODFW staff notes from the time reported high sedimentation, many empty red abalone shells washing up, and a lack of live red abalone in areas where qualitative surveys had documented them prior (Lukas[13]).

Next, from 2014 to 2017, a series of events caused another major reduction in red abalone abundance in Oregon. Bull kelp beds during this period were dramatically reduced across the region due to intense ocean warming, loss of sea stars (via sea star wasting syndrome), and dramatic increases in sea urchin populations (Rogers-Bennett and Catton, 2019). The loss of bull kelp beds resulted in abalone starvation causing once again high numbers of empty abalone shells found on Oregon beaches (personal observ.). Survey data also shows substantial reductions in red abalone populations, where low densities found in 2015 were reduced even further, to around half what they were in previous work (Table 3).

Flat Abalone

Along the U.S. west coast, flat abalone are considered a minor species in the context of fisheries due to their small size and sparse abundance. Southern Oregon is a rare stronghold for flat abalone, a species uncommon in California (Rogers-Bennett, 2007) and largely absent in Washington (Straus, 2010). While flat abalone were not a target of a recreational fishery in Oregon, they were the target of a commercial fishery.

[16]Golden, J., and J. Martin. 1995. Abalone and salmon culture at former Oregon Aqua site. ODFW unpubl. memo, 22 p.

[17]Langdon, C. 2019. Oregon State University, Corvallis. Personal commun.

[18]Foley, T. 2019. Oregon commercial sea urchin diver. Personal commun.

Commercial Fisheries, 2001–08

Fishery Management

Oregon's commercial fishery for flat abalone began in 2001, and in 2008 it was closed amid sustainability concerns. The fishery was managed by: 1) limited entry (one permit), 2) an annual quota, and 3) minimum legal size (MLS). The permittee was required to keep logbooks, allow access to catch for measurement, and perform pre- and post-fishing surveys. Quota was established at 1,361 kg (3,000 lb) per year in 2001, then reduced to 907 kg (2,000 lb) per year in 2008, the final year of the fishery. A MLS was set at 114 mm (4.5 in) SL in 2001, then increased in 2008 to 121 mm (4.75 in).

Logbooks and landing receipts were required. The 2001–08 commercial flat abalone fishery logbooks included date, catch area, dive time, mean depth, and estimated weight of catch. Landing receipts recorded date, port, whole weight, and value.

In each year of the fishery, the quota was met, amounting to total landings of 10,092 kg (22,249 lb) of flat abalone from 2001 to 2008. By port, fishing was focused on south coast ports with 44% landed in Port Orford and 41% landed in Gold Beach. The remaining landings occurred at Brookings, Charleston, and Depoe Bay (8.2%, 5.7%, and 1.1%, respectively). By catch area, focus was on Oregon's two large offshore reefs, Rogue Reef (43.4% of catch) near Gold Beach and Orford Reef (20.2%) but also included high catch from Nellies Cove (22.1%), a small protected cove near Port Orford. The remaining 14.3% of catch was from other areas along the south coast but also with some catch in the central coast near Depoe Bay (Fig. 7). Fishery metrics showed "fishing down" effects. Fishing time became longer (Fig. 8a), depth of catch became deeper (Fig. 8b), and CPUE decreased (Fig. 8c).

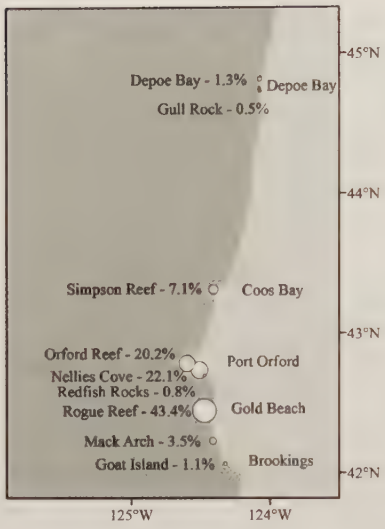


Figure 7.—Catch in Oregon's commercial flat abalone fishery by reef area on the south coast from 2001 to 2008.

Fishery Management

Market sampling was conducted sporadically. Fishery deliveries were selected haphazardly, then 50 flat abalone per delivery were selected randomly and measured with 300 mm Vernier calipers. Shell length was measured for each flat abalone; whole weight, sex, and gonad condition data were occasionally included. Market samples showed a 54:46 male to female sex ratio for the 1,443 flat abalone where sex was determined. Overall, there was little variation in mean size by area or time, typically remaining close to the minimum size length in the rules of the fishery (Table 5).

To establish a length to weight relationship for flat abalone, shell length (SL) of flat abalone from catch samples were converted to their biomass equivalents (W) (based on wet weight of all individually weighed samples collected between 2004 and 2008, R^2 =0.87, n=937) using this allometric relationship: $W = 0.0000928 \times SL^{3.0852}$ (Fig. 9).

Surveys

Two survey efforts have quantified flat abalone populations at key areas: 1) fishery dependent surveys conducted during the 2001–08 commercial fishery (within fishing areas) and 2) fishery independent surveys of 2015 and 2019 at two sites (Port Orford and Brookings).

Fishery Dependent Surveys (2001–08)

Surveys for flat abalone were performed (by the permittee) each time fishing began in a new area (2001–02),

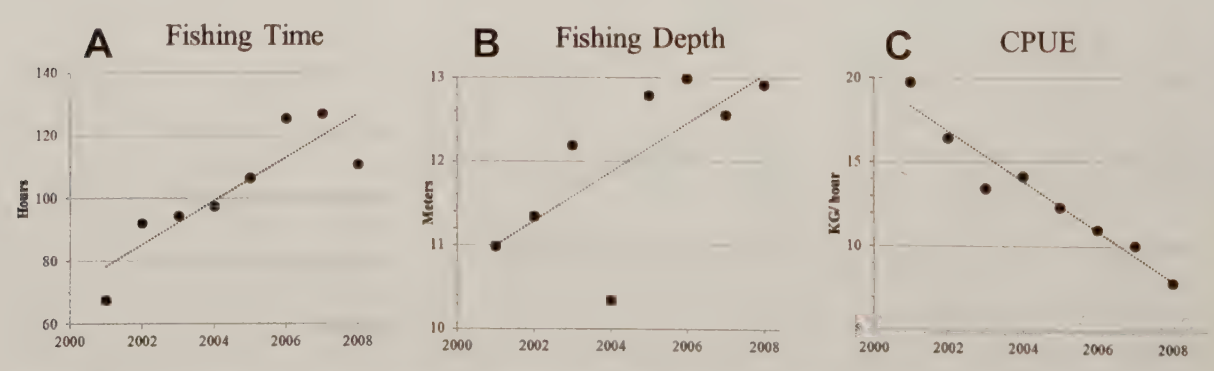


Figure 8.—Fishery metrics of Oregon's commercial flat abalone fishery from 2001 to 2008, by year. A) fishing time (hours) and B) fishing depth (m) showed increasing trends while C) fishery CPUE (catch (kg)/ fishing time) had a decreasing trend.

Table 5.—Mean shell length (mm) and count of measurements (in parentheses) from flat abalone market samples from 2001 to 2008 in Oregon.

	Area						
Year	Rogue Reef	Nellies Cove	Orford Reef	Depoe Bay	Goat Island	Gull Rock	Total
2001	131.6 (10)				146.9 (25)	142.0 (30)	142.3 (65)
2004	134.8 (109)	129.3 (38)					133.4 (147)
2005	128.7 (236)		121.5 (70)				127.1 (306)
2006	129.8 (141)		127.9 (103)				129.0 (244)
2007			126.9 (259)				126.9 (259)
2008	137.6 (268)	130.0 (113)	130.2 (257)	138.0 (228)			134.5 (866)
Avg.	133.0 (764)	129.8 (151)	127.7 (689)	138.0 (228)	146.9 (25)	142.0 (30)	131.7 (1,887)

then revisited approximately 5 years later (2006–08). Survey sites were typically within kelp beds, in highly rugose habitats and at depths of 5–15 m. A 100 m transect was surveyed, where in each 10 m interval a 1-m^2 quadrat was evaluated for count and size of flat abalone. This methodology is strongly biased towards "hot spots"; however, it could serve as a maximum abundance survey or for temporal comparison.

Surveys conducted prior to fishing (2001–02) showed robust populations of flat abalone. Follow-up surveys (2006–08) showed substantial "fishing down" of those same populations. Densities at the primary catch areas (Rogue Reef, Orford Reef, and Nellies Cove) declined by 59%, 28%, and 80%, respectively (Table 6). Survey methodology (particularly its fishery dependence) was not ideal; however, they were performed carefully and directly in cooperation with ODFW. This was necessary because ODFW did not have a diving program after 1999. The declines in density at each catch site were expected and, despite suboptimal methodology, agree with other fishery metrics (i.e., changes to effort, depth, and CPUE (Fig. 7)).

Fishery Independent Surveys in 2015 and 2019

Two efforts to quantify flat abalone abundance occurred in areas where flat abalone are historically common. Quantitative surveys for abalones occurred at Port Orford and Brookings (methodology described in red abalone section) in 2015 and 2019. The results of these surveys show two things: 1) the overall very low density of abalones in Oregon and 2) the significant

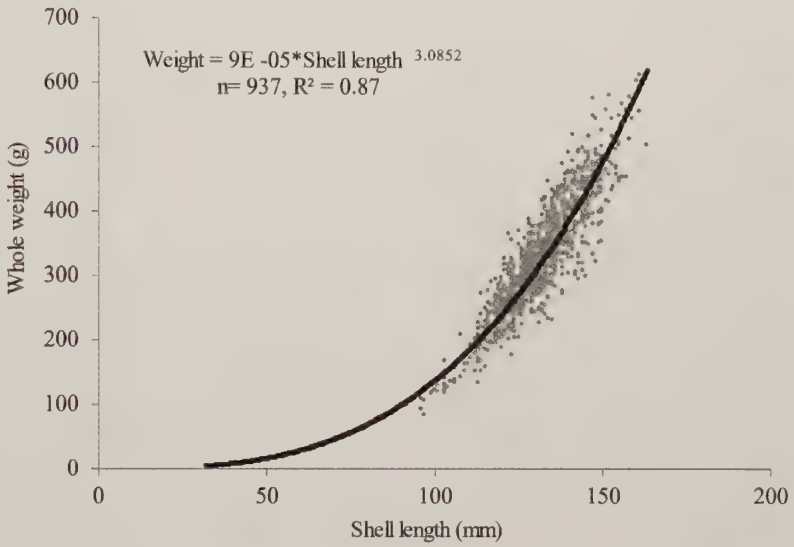


Figure 9.—Length/weight relationship of individually measured flat abalone market samples (n=937) from 2001 to 2008 in Oregon.

decrease ($p<0.01$, MW test) of flat abalone between 2015 and 2019 (Table 7).

The flat abalone abundance was reduced sequentially, likely reacting to fishing, then poor environmental conditions. An important example of Oregon's flat abalone population trajectory is Nellies Cove, an area which has historically had a robust population. Densities found in fishery dependent surveys of 2001 were 5.6 flat abalone/m^2. By 2008, flat abalone density reduced to 1.1/m^2.

Next, fishery independent surveys showed a low density of flat abalone in 2015 (0.013/m^2). By 2019, density reduced to 0.002/m^2; only one flat abalone was found within 11 belt transects, covering 660 m^2 of high quality abalone habitat. The reduction in abundance of this once prominent species appears nearly complete, where a species once found in commercial quantities could become locally extinct.

Pinto Abalone

Pinto abalones range from southeast Alaska to Baja California, Mexico, encompassing Oregon. Despite being central to its range, pinto abalone are extremely rare in Oregon. Although targeted searches using methods known to detect pinto abalone were preformed, no live specimens were found in Oregon for many years (Marriage[4]; Reimers[19]), includ-

[19]Reimers, P., and D. Snow. 1975. Rare and endangered species. ODFW unpubl. memo.

Table 6.—Density of flat abalone (per m^2) from fishery dependent surveys (2001–08) in Oregon.

	Area								
Survey	Rogue Reef	Nellies Cove	Orford Reef	Depoe Bay	Mack Arch	Goat Island	Redfish Rocks	Simpson Reef	Average
Baseline	2.2	5.6	3.2	1.1	3.1	1.8	2.3	2.5	2.7
Post-harvest	0.9	1.1	2.3						1.4
Average	1.8	3.4	2.7	1.1	3.1	1.8	2.3	2.5	2.4

Table 7.—Densities of flat and pinto abalone from fishery independent surveys (2015 and 2019) at Nellies Cove, Port Orford, Oreg.

	Densities (per m^2)		
Year	Flat	Pinto	Surveyed area
2015	0.013	None	900
2019	0.002	0.014	660

ing more than 50 years of ODFW dive surveys.

In 2009, the first known live pinto abalone was discovered in Oregon, at Orford Reef by sea urchin divers. Since 2009, pinto abalone have been found more frequently, noted as seen by sea urchin divers a few times a year (Ashmon[20]). In 2015, surveys specifically targeting high quality abalone habitats found no pinto abalone; however, in 2019, nine pinto abalone were found (personal observ.).

Black Abalone

Black abalone have an ambiguous record of presence within Oregon. Early publications identify Coos Bay as its northern range extent (Bonnot, 1948; Cox, 1962; Marriage[4]), citing a general book on marine shells of North America (Oldroyd, 1927). Oldroyd lists the range of black abalone as extending to Coos Bay, although the furthest north a museum specimen has been collected is Crescent City, Calif. (Geiger, 1999). Oldroyd referred to two earlier documents (Leach, 1814; Reeve, 1845) when describing black abalone, neither of which included specific geographic range information. It appears likely Oldroyd's geographical reference was from field observations of the time; however, these were not described specifically. Recently, when black abalone were listed as "endangered" under the ESA, this ambiguous nature of their history in Oregon was highlighted (Butler, et al., 2009:16–17).

A thorough internal ODFW historical document review may elucidate the issue of black abalone presence in Oregon. OFC biologist Lowell Marriage describes his personal finding of live black abalone one mile south of Coos Bay in 1952 and 1954 letters to colleagues (Marriage[4, 21]). Considering the expertise of Marriage, we believe this finding to be accurate. In addition, a University of Oregon biology professor offered some evidence of black abalone presence, though the findings appeared inconclusive (McConnaughey[22]). Over many years, ODFW performed different types of surveys in suitable habitats; however, to our knowledge, no black abalone have been found since.

Discussion

Changes in Abundance

Over a relatively short period, shifts in abalone range and abundance have occurred in Oregon. Populations of each of the four abalone species occurring in Oregon have not been stable.

Red abalone population levels have varied substantially over time. While populations were initially thought to be robust enough to support a commercial fishery (1950's), by the 1960's and 1970's they were difficult to find (evidenced in biologist notations and low success in the recreational fishery). By the 1990's and 2000's they were easily found in the recreational fishery, being taken at high rates and even intertidally. Quantitative surveys of the 2010's showed a low number of red abalone, then a marked decrease was found in 2019 and 2022 surveys.

It is important to note that both major decreases (1960's and 2010's) appeared tied to environmental events (tsunami/floods of 1964 and environmental conditions of late 2010's). At these two times, empty (dead) red abalone shells were reported en masse at Oregon beaches, particularly noteworthy given how rare they are. The causative relationship of these environmental events to the declines of red abalone are difficult to assess. More robust and frequent fishery independent surveys are needed to properly understand trends and mechanisms. The future of red abalone populations appears to be in jeopardy; however, the species did make a comeback from low levels (albeit unquantified) following the 1964 environmental events.

Flat abalone were noted commonly throughout qualitative red abalone surveys in the 1950's–70's. In the late 1990's interest by a sea urchin diver indicated abundances were high enough to make a commercial fishery viable, which occurred from 2001 to 2008. As the fishery progressed, abundance declined rapidly. One site that was typically a stronghold of flat abalone, Nellies Cove (near Port Orford), experienced an 82% reduction in abundance from 2001 to 2007, likely corresponding to fishing. Subsequent fishery independent surveys showed further reduced populations; in 2015 (Groth[1]), then in 2019 only one flat abalone was found during a survey (personal observ.) of a site which in the past had yielded catch rates of more than 100 flat abalone per hour during the early years of the commercial flat abalone fishery. A recent and dramatic decline in flat aba-

[20]Ashmon, M. 2019. Oregon commercial sea urchin diver. Personal commun.

[21]Marriage, L. D. 1952. Presence of abalone in Oregon. OFC unpubl. memo.

[22]McConnaughey, B. H., Univ. Oreg. 1952. Letter regarding abalone distribution in Oregon.

lone has also been noted in California populations (Rogers-Bennett[23]).

Pinto abalone populations have also changed. After years of exploration and surveys, no live specimens had been documented in Oregon until 2009 when a few were found (personal observ.). Since 2009, they appear to be more prevalent, even to be quantifiably detected in 2019 surveys, when a few individuals were found.

Lastly, black abalone, a species where individuals were once found in Oregon, has not reappeared since findings of the 1950's (Marriage[4, 21]; McConnaughey[22]). This past discovery and disappearance; however, may give insight into the ephemeral nature of this species group beyond its range extents.

Red Abalone Size in Oregon

It is common for this species to grow to its largest individual size in the coldest environments within their range (Bergmann, 1847; Vinarski, 2014). Oregon's population of red abalone are well characterized by this biological rule as they are uncommon, but grow to the species largest sizes at their polar range extent. Accordingly, Oregon's red abalone follow this model.

The largest abalone shells in the world have been found in Oregon. Most famously, a 313 mm (12.32 inch) red abalone, frequently cited as the world record abalone shell, is widely reported to be Oregon taken both by these authors many personal communications and by online sources (Tissot[24]). Less famously, ODFW records show at least two other red abalone shells found in Oregon have been larger.

1) ODFW staff measured a 318 mm (12.5 inch) red abalone during the 1958–62 exploratory commercial fishery (Snow[6]), probably from the Brookings area.
2) In 1972, ODFW staff recovered another 318 mm red abalone (shell) near Port Orford (unpubl. data, ODFW).

The large individual size of red abalone presents a rare situation in marine invertebrate fisheries, where trophy shell catch, rather than food sourcing, has motivated most fishery participants. In recent years, as daily and annual limits for red abalone have declined, divers have focused on catching the greatest individually sized red abalone (Tissot[24]). Incorporating the unique trophy size of Oregon's red abalone could be used as a management tool to increase the fisheries sustainability by narrowing fishery selectivity and reducing total catch.

Oregon's Abalone Future

Worldwide and regionally, abalone fisheries have been scrutinized closely given frequent failures (Vileisis, 2020). As of 2018, all Oregon abalone fisheries have been closed or suspended due to sustainability concerns. While fisheries are valuable in developing societal investment in resources and contribute to the local economy, sustainability must be achieved to allow both to persist. For abalones, sustainability has proven hard to achieve. Given their paucity, fisheries for flat, pinto, and black abalone in Oregon appear unlikely to be sustainable at any level; however, the high value of large-sized red abalone may be particularly useful to management.

The recreational red abalone fishery in Oregon persisted for many years, ultimately closing due in largest part to environmental conditions, not overfishing. Despite the many years of fishing, catch rates were highest in the most recent years. The exceptionally large size of red abalone in Oregon allows fishery participation to be valuable even at extremely low levels of catch.

Improving the sustainability of Oregon's recreational red abalone fishery would require better population data and the development of appropriate targets and limits. Few quantitative surveys have occurred; these have allowed a window into density trends but little more. More survey data are needed to understand: 1) the number of surveys required to appropriately assess population levels, 2) the range and species diversity of abalones in Oregon, and 3) the abundance of abalone relative to restoration needs and fisheries allotments.

Modern abalone management uses density metrics; however, being at the edge of their range, densities in Oregon are likely to never meet the minimums of other areas where durable fisheries have occurred. Choices for the future could include: 1) no fisheries for red abalone, 2) limited open access fisheries, or 3) highly limited "big game" style fisheries. While the choice of "no fisheries" is appealing and appropriate from an immediate conservation point of view, long term conservation must consider societal investment. For abalones this investment appears strongest among fishery participants. Developing small fisheries, such as a trophy fishery may be possible given Oregon's unique red abalone population and could retain the societal investment critical to conserve stocks of red abalone.

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[23]Rogers-Bennett, L. 2021. California Department of Fish and Wildlife. Personal commun.

[24]Tissot, B. 2014. The hunt for monster red abalone, accessed April 2, 2023 (avail. at https://briantissot.com/2014/09/07/the-hunt-for-monster-red-abalone).

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Fish Bycatch in the North Pacific Halibut, *Hippoglossus stenolepis,* and Groundfish Fisheries

DAVID WITHERELL and MICHAEL FEY

Introduction

Unwanted fish are taken incidentally in almost every U.S. commercial and recreational fishery. Fish are discarded because they are prohibited from being retained by regulation (for example, restrictions on the amount, size, sex, or species of the catch), or are simply not wanted due to the fish being of an unpreferable size (typically too small), or a species that may be less desirable to retain, or a lower value or unmarketable species taken in commercial fisheries. Regardless of the reason that fish are discarded, it is all considered bycatch, except for fish released alive under a recreational catch and release fishery management program. Bycatch can be a social and management issue concerning waste of natural resources or an allocation issue among different fishery sectors. Bycatch can also be a conservation issue due to impacts on biodiversity and fish populations, particularly if the bycatch is unmonitored and unregulated.

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In the United States, the primary law for management of federal marine fisheries is the Magnuson Stevens Fishery Conservation and Management Act (MSA) of 2007. The MSA establishes ten National Standards for the conservation and management of fisheries and requires that any plan or regulation developed must be consistent with these standards. However, not all of the National Standards can be met simultaneously, so each management decision must balance the tradeoffs among the standards. The overriding factor is National Standard 1, which requires that conservation and management measures shall prevent overfishing while achieving, on a continuing basis, the optimum yield (OY) from each fishery for the United States fishing industry. National Standards 2 through 10 provide further requirements for conservation and management measures.

Bycatch is addressed by National Standard 9, which requires that conservation and management measures shall, to the extent practicable, minimize bycatch. Further, the standard requires that the mortality of any bycatch that cannot be avoided must be minimized. The MSA defines bycatch as "those fish which are harvested in a fishery, but which are not sold or kept for personal use, and includes economic discards and regulatory discards. The term "bycatch" does not include fish released alive under a recreational catch and release fishery management program." The term "regulatory discards" means "fish harvested in a fishery which fishermen are required by regulation to discard whenever caught, or are required by regulation to retain, but not sell." The MSA further defines fish to be finfish, mollusks, crustaceans, and all other forms of marine animal and plant life other than marine mammals and birds.

The MSA also established eight regional fisheries management councils to develop fishery management plans (FMP's) and fishery regulations for their respective regions, based on the National Standards and other provisions for conserving and managing fisheries, including minimizing bycatch. The North Pacific Fishery Management Council (Council) was given the authority to develop FMP's and federal fishery regulations, subject to approval by NOAA's National Marine Fisheries Service (NMFS), for fisher-

ABSTRACT—Fish caught but not retained (i.e., fish that are discarded) during fishing operations are considered bycatch. In the commercial fisheries of the North Pacific, fish are discarded because they are either unmarketable species and unprofitable to retain or process or they are required by regulation to be discarded. Catch and bycatch in the Pacific halibut, Hippoglossus stenolepis, *and groundfish fisheries are closely monitored and recorded by observers at sea or shoreside, or by cameras onboard fishing vessels, to ensure that bycatch can be accurately estimated for individual species across the different fisheries and management areas.*

Overall North Pacific bycatch is relatively low compared to other commercial groundfish fisheries in other areas. Of the 2,058,816 t of fish caught in the North Pacific halibut and groundfish fisheries in 2021, 97,083 t (4.7%) were discarded. Bycatch accrues towards the annual catch limits for groundfish stocks, and thus has not been a conservation concern for most species.

Nevertheless, bycatch management can be a very contentious issue, given the allocative aspects for any species taken that is targeted by another gear type, and the socioeconomic and cultural concerns of Alaska Native and rural communities regarding bycatch of Chinook salmon, Oncorhynchus tshawytscha, *and chum salmon,* O. keta. *In this paper we review the data on bycatch of fish in the North Pacific halibut and groundfish fisheries and review the management measures that have been implemented to control and minimize bycatch to the extent practicable.*

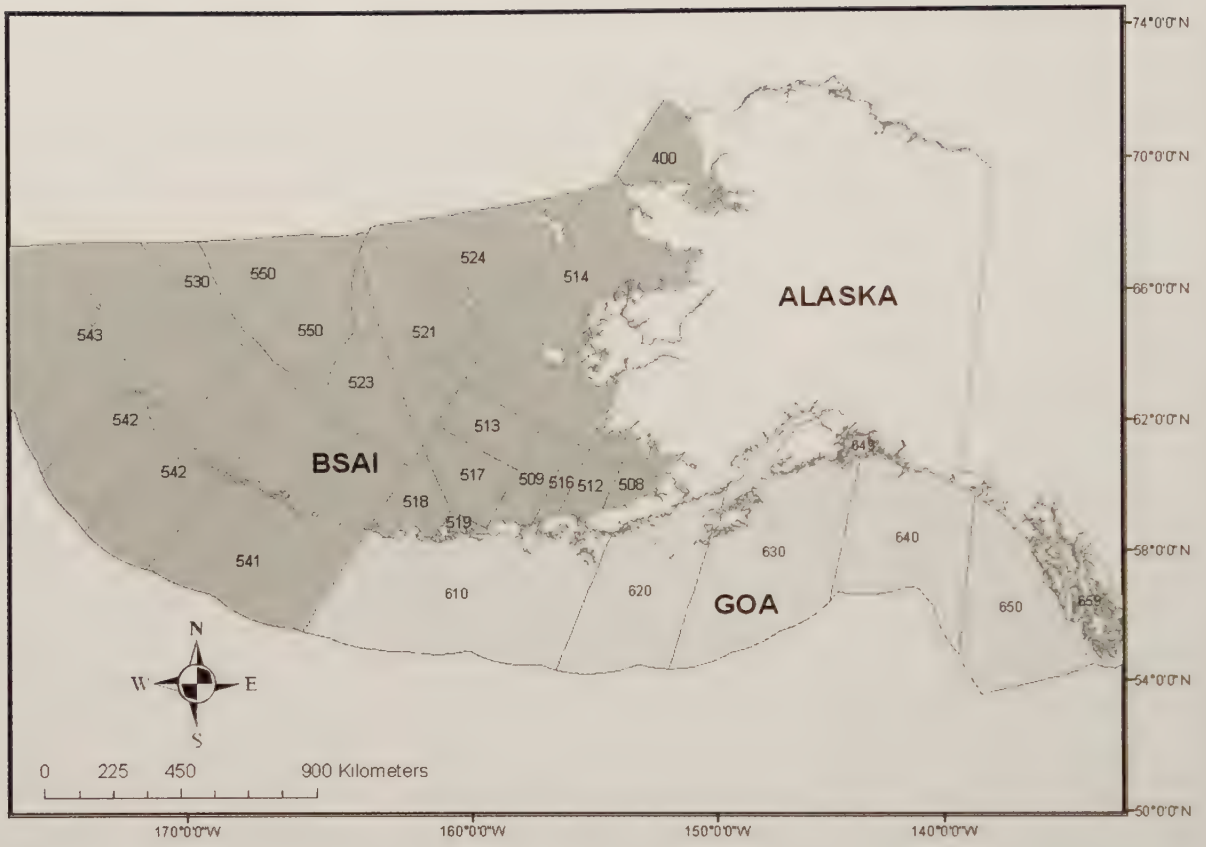

Figure 1.—Management and statistical areas for federal groundfish in the North Pacific: GOA (Gulf of Alaska areas 610-650) and BSAI (Bering Sea areas 508–530, Aleutian Islands areas 541–543).

ies in the Exclusive Economic Zone (3–200 n.mi.) off Alaska.

There are two FMP's that guide the management of groundfish fisheries in the North Pacific: the FMP for Groundfish of the Bering Sea and Aleutian Islands Management Area (BSAI Groundfish FMP), and the FMP for Groundfish of the Gulf of Alaska (GOA Groundfish FMP). These management areas, and the statistical areas within each that are used to manage catch and bycatch, are shown in Figure 1.

Pacific halibut, *Hippoglossus stenolepis*, is managed by the International Pacific Halibut Commission (IPHC) and through regulations implementing the Northern Pacific Halibut Act of 1982 (Halibut Act). The Halibut Act, at section 773c, also provides the Council with authority to develop regulations that are in addition to, and not in conflict with, approved IPHC regulations. The IPHC is authorized to conserve the halibut stocks and allocate the available catch limit between the United States and Canada. The Council has the authority to manage and regulate halibut fisheries in the Exclusive Economic Zone off Alaska, including allocating the harvest and fishing privileges among U.S. fishermen, subject to approval by the Secretary of Commerce. This authority also extends to regulating bycatch in the halibut fisheries.

In 2002, an environmental organization petitioned NMFS to count, cap, and control bycatch, and asserted that the agency was not complying with its obligations to minimize bycatch (Dobrzynski et al., 2002). In response to this petition, NMFS developed a National Bycatch Strategy, thus elevating the importance of the issue and committing the agency to expend additional effort to minimize bycatch (Benaka and Dobrzynski, 2004).

In 2011, NMFS issued its first national bycatch report in an attempt to quantify bycatch across different U.S. fisheries and regions based on 2005 data for those fisheries where information was available (Karp et al., 2011). The bycatch report has since been updated several times, with the latest report based on data through 2015 (Benaka et al., 2019).

This paper reviews the bycatch (as defined by the MSA) of fish taken in

the different federally managed fisheries in the North Pacific, the management measures applied to minimize bycatch to the extent practicable, and discusses ongoing challenges in the management of bycatch in the North Pacific halibut and groundfish fisheries.

Background

North Pacific Fisheries

The federally managed groundfish fisheries in the North Pacific target an array of commercially important fish stocks including walleye pollock, *Gadus chalcogrammus;* Pacific cod, *Gadus macrophalus*; sablefish, *Anoplopoma fimbria*; yellowfin sole, *Limanda aspera*; northern rock sole, *Lepidopsetta polyxystra*; and other flatfish species; Pacific ocean perch *Sebastes alutus*; and other rockfish, *Sebastes* spp. and *Sebastolobus* spp.; and Atka mackerel, *Pleurogrammus monopterygius* (Witherell and Armstrong, 2015). The fisheries are managed using an ecosystem-based approach to achieve sustainable fisheries that provide benefits for harvesters, processors, recreational and subsistence users, and fishing communities. The fisheries 1) are maintained by healthy, productive, biodiverse, resilient marine ecosystems that support a range of services; 2) support robust populations of marine species at all trophic levels, including marine mammals and seabirds; and 3) are managed using a precautionary, transparent, and inclusive process that allows for analyses of tradeoffs, accounts for changing conditions, and mitigates threats (NPFMC, 2014a).

Management measures implemented to achieve sustainability include annual catch limits, which are established for each species or species group, and are set at conservative levels that are at or below the acceptable biological catch level and below the overfishing threshold corresponding to F_{MSY} (DiCosimo et al., 2010). In addition to catch limits, the fisheries are managed using traditional tools including: licenses and permits, allocation of catch limits among sectors, seasons and seasonal allocations, limited access privilege programs such as individual fishing quota (IFQ) and catch share cooperatives, monitoring and reporting requirements, an overall optimum yield limit on how much groundfish can be harvested annually from the ecosystem, limits on bycatch of non-target species, a prohibition on targeting forage species, gear requirements to reduce bycatch or habitat impacts, conservation areas, and measures to minimize fishery impacts on marine mammals and seabirds (Witherell et al., 2000; Witherell and Woodby, 2005; Heltzel et al., 2011; Fina, 2011).

The groundfish fisheries are annually prosecuted by about 400 vessels, with the different fleets (i.e., sectors) categorized by gear type (trawl, pot, longline, jig) and mode of operation (vessels that catch and deliver to processing facilities or to vessels that catch and process at sea) (Witherell et al., 2012). For example, walleye pollock is targeted using pelagic trawl gear by catcher/processors operating in the BSAI and catcher vessels delivering to shoreside and vessel-based processing facilities in both the BSAI and GOA areas. Pacific cod is targeted by vessels fishing with bottom trawl, pot, and longline gears that either process at sea or deliver shoreside. Rockfish and most flatfish are targeted by catcher/processors using bottom trawls in the BSAI and western GOA, but these stocks are targeted only by catcher vessels delivering shoreside in the central GOA. A few vessels target Pacific cod (and to a lesser extent rockfish) with jig gear.

The commercial halibut fishery off Alaska targets halibut with longlines, although pot gear has recently been authorized for this species. The fishery operates through annual issuance of IFQ to quota share-holders. About 1,000 vessels participate in the commercial halibut fishery each year, and about one-third of the vessels also participate in the sablefish IFQ fishery using longlines and pots (Witherell et al., 2012; Szymkowiak et al., 2020). Commercial halibut catch limits for each IPHC management area are established annually by the IPHC, after accounting for projected removals from the subsistence, charter, and sport fish sectors, and the estimated removals of halibut bycatch mortality in the halibut and groundfish fisheries. NMFS then issues catch limits for each management area as quota (in pounds) to individual quota holders to harvest. A regulatory minimum size limit of 81.3 cm (32 in) has been in place since 1973, and all halibut under this size must be discarded in the halibut directed fishery. Keith et al. (2014) provides an overview of conservation and management measures for the halibut fishery.

Observer Program

A comprehensive observer program, combined with real time reporting and management, as well as strict enforcement of regulations and binding civil agreements among harvesters in cooperatives, ensures that conservation and management measures are effective. In the groundfish and Pacific halibut fisheries, catch and discard data are collected through the North Pacific Observer Program. The program provides the regulatory framework to deploy observers and electronic monitoring (EM) systems to collect data necessary for the conservation, management, and scientific understanding of the commercial fisheries in the BSAI and GOA management areas. In the North Pacific fisheries, EM systems use video technology to collect data on catch and discard, and provide monitoring for compliance with regulations.

Data collection through the Observer Program provides a reliable and verifiable method for NMFS to collect fishery-dependent catch data, biological information on fish, and data on seabird and marine mammal interactions with fisheries. Observers and EM systems collect information on catch and discards (including inadvertent discards such as fish that drop off hooks and fish that are bled from trawl codends) onboard fishing vessels and at onshore processing plants (AFSC, 2021). These data are used in the Catch Accounting System to ex-

trapolate and estimate total catch and discards in the fisheries (Cahalan et al., 2014) and enable real-time catch information for in-season management of the fisheries.

The monitoring program is based on a statistically rigorous sampling design (Cahalan and Faunce, 2020). Vessels fall into two categories of coverage, a full coverage category and a partial coverage category. In the full coverage category, vessels must have one or two observers onboard on every trip, and vessels obtain observers by contracting directly with certified observer providers. Vessels in this category include catcher/processors as well as catcher vessels that are participating in specific limited access privilege program fisheries. The vast majority of the groundfish catch falls under the full coverage category, and over 93% of the trawl catch is on trips with full coverage (NMFS, 2022).

A small portion of the total groundfish catch and all directed halibut catch falls under the partial coverage category (vessels take observers when randomly selected but not on every trip), which is funded by revenue generated through an ex-vessel value-based fee (1.65% of the ex-vessel price of the landed catch for catches not landed in the full coverage category), that is split between the vessels and the processors. Vessels in the partial coverage category currently fall into four different pools that determine their monitoring coverage: the no selection pool (vessels using jig gear and vessels <12.2 m in length using pot or longline gear; these vessels have no probability of being selected for monitoring); the trawl EM pool (specific pollock catcher vessels using pelagic trawl gear); the fixed-gear EM pool (non-trawl vessels that chose to use EM and are approved by NMFS); and the observer trip selection pool (catcher vessels ≥ 12.2 m (40 ft) in length that are not in the EM pool).

To ensure that statistically reliable data collection occurs in the partial coverage category with available fee revenue, NMFS prepares an annual deployment plan to deploy at-sea and shoreside fishery observers and EM systems to processing plants and vessels participating in halibut and groundfish fisheries (NMFS, 2021). EM is a growing component of this plan. Vessels in the trawl EM pool are required to have cameras operating at all times to verify compliance with maximized retention requirements. Catch accounting for the vessel's catch and bycatch is done through industry reports and shoreside plant observers. Vessels in the fixed-gear EM pool turn on their cameras during randomly selected trips and the EM information is used for catch and bycatch estimation. In the observer pool, trips are randomly selected for onboard observer coverage.

Together, monitoring of the vessels in the full coverage and partial coverage categories provides a comprehensive data collection program from which to extrapolate and provide accurate estimates of discards. In 2021, for example, 99% of the catch (including discards) in the BSAI halibut and groundfish fisheries, and 46% of the catch in GOA was monitored by an at-sea or shoreside observer and/or by EM (NMFS, 2022). In the 2021 GOA fisheries, the percent of total discards on trips monitored by observers or by EM where some video was reviewed or sampled by observers shoreside, was as follows: 20% for pot gear, 21% for hook and line gear, 48% for pelagic trawl gear, and 91% for bottom trawl gear (NMFS, 2022). In the 2021 BSAI fisheries, 21% of the discard was on monitored trips for pot gear, 84% for hook and line gear, 100% for pelagic trawl gear, and 99% for bottom trawl gear (NMFS, 2022). Bycatch estimation for North Pacific groundfish fisheries received high scores in a classification system that considers the quality of bycatch data and the reliability of bycatch estimation methods (Benaka et al., 2021).

Types of Bycatch and Discards

Bycatch and discards are fully accounted for in the North Pacific, and bycatch can be categorized into different types based on why the fish are discarded. The categories are economic discards and regulatory discards. A summary of these types of discards in the North Pacific groundfish and Pacific halibut fisheries is provided below.

Economic discards of groundfish or other species may occur for various reasons including fish quality, fish size, and marketability of the species in general. At the current time, many species of fish and invertebrates caught in the North Pacific do not have markets and thus processors will not pay a fisherman any price to land them. Examples of unmarketable fish include sculpins, grenadiers, and sharks, as well as invertebrates such as jellies and sea stars. Additionally, fisherman may choose to discard fish due to having limited hold or freezer space, or even to prevent one species from degrading the quality of other species in the fish hold. Discarding may also occur for fish of unmarketable quality due to wounds from marine mammal bites or parasites, sand fleas, or damage that occurred during capture, or when a vessel does not deliver in sufficient time and the fish is no longer of a marketable quality. It is therefore understood that any species, regardless of its value, could end up discarded due to economic reasons.

The second type of bycatch is regulatory discards, which occur due to size limitations, attaining the catch limit within a season or year, catching more halibut or sablefish than the available IFQ held by those on the vessel, exceeding the trip limit for pollock in the western GOA, or attaining Maximum Retainable Amount (MRA) limits during a fishing trip. MRA's are established by regulation for each individual groundfish catch species or complex, as a percentage of the retained catch of a species that is open for directed fishing (i.e., the basis species). A vessel participating in directed fishing can only retain up to the MRA for species closed for directed fishing, and all catch of that species exceeding the MRA must be discarded. The intent of MRA's is to provide an incentive to avoid targeting a species that is closed to directed fish-

ing when fishing for other species that are open to directed fishing (Ackley and Heifetz, 2001). NMFS can also require all catch of a stock to be discarded if the total catch of that stock is getting close to attaining the annual catch limit for that stock, thus removing any incentive to further target that species or stock. Size limits also create regulatory discards. While there are no size limits for any groundfish species, IPHC regulations dictate that any Pacific halibut harvested in the commercial halibut directed fishery must be at least 81.3 cm (32 in) long, so halibut smaller than this minimum size are discarded by regulation (Stewart et al., 2020).

Another type of regulatory discard that is specific to the groundfish fisheries in the North Pacific is called prohibited species catch (PSC). Prior to the passage of the MSA in 1976, the principal domestic fisheries in Alaska were for Pacific salmon, red king crab, Pacific herring, and Pacific halibut. These principal species were given special status as "prohibited species" in the BSAI and GOA Groundfish FMP's that were implemented in the early 1980's, meaning that these species could not be retained in the groundfish fisheries. Regulations require all PSC species caught in groundfish fisheries to be released immediately with a minimum of injury. Bycatch limits for PSC species (red king crab, *Paralithodes camtschaticus*; Tanner crab, *Chionoecetes bairdi*; snow crab, *C. opilio*; Pacific herring, *Clupea pallasii*; Pacific halibut; Chinook salmon, *Oncorhynchus tshawytscha*; and chum salmon *O. keta*) were established for groundfish trawl fisheries to reduce the impacts on these species that were traditionally harvested by other gear types (Witherell and Pautzke, 1997).

Halibut bycatch is typically managed in terms of total mortality of discards. To calculate total mortality, specific discard mortality rates are applied to total halibut discards in different groundfish fisheries (set by gear type) and the halibut fishery based on scientific studies of halibut survival as indicated by the viability of Pacific halibut at the time of discard (Kaimmer and Trumble, 1998; Armstrong et al., 2016). These discard mortality rates are updated every few years based on updated viability observations from at-sea observers. For 2021, for example, BSAI halibut discard mortality rates applied to specific gear types and groundfish sectors were as follows: 100% for Pacific halibut caught with pelagic trawls, 84% caught with bottom trawls on mothership and catcher processors (on hauls not subject to deck sorting), 59% caught with bottom trawls on catcher vessels, 9% with longline gear, and 32% with pot gear. The IPHC applies a 16% mortality rate to Pacific halibut discarded in the halibut longline fishery for use in stock assessment and management (Stewart and Webster, 2020).

Bycatch Management Measures

The Council works closely with NMFS to meet the objectives of the National Bycatch Strategy and standards for effective management of bycatch in the North Pacific, including monitoring and quantifying fisheries bycatch, encouraging research to minimize bycatch, assessing the effects of fishing on bycatch populations, setting operational objectives consistent with legal requirements and policies, considering the cost effectiveness of mitigation measures, considering enforcement of the management measures, evaluating the effectiveness of bycatch measures, and regular communication with stakeholders on bycatch management actions (Benaka and Dobrzynski, 2004; Kirby and Ward, 2014). This process allows management measures being developed to meet the MSA National Standards, be transparent to stakeholders, and effective. The Council, NMFS, and participants in the halibut and groundfish fisheries have utilized many of the approaches identified by Hall and Mainprize (2005) to minimize bycatch, including the use of regulations, societal pressure, encouraging the fishery sector to find solutions that do not require regulation, and technological solutions.

Regulatory approaches include establishment of bycatch limits for many PSC species to limit the incidental catch of these species in the groundfish fisheries. These PSC limits were established to address concerns of fishermen, who target a particular species, to seek limits on the amount of that species caught incidentally in the halibut and groundfish fisheries. Most of the PSC limits were established based on historical bycatch amounts and set at amounts expected to have limited impacts (1% or so) on the population of PSC species (Witherell and Pautzke, 1997).

Interest in reducing bycatch of these species increases during times when the biomass of these species declines and bycatch represents a higher portion of total removals. This is particularly the case for Pacific halibut and salmon PSC where annual PSC limits, until recently, have not been directly tied to stock abundance (Witherell et al., 2002; Stram and Ianelli, 2014; DiCosimo et al., 2015; NPFMC, 2021). Total PSC limits for BSAI groundfish trawl fisheries, using 2021 as an example, included 2,805 t of halibut mortality (and 710 t for longline fisheries), 2,723 t of herring, 80,161 Bristol Bay red king crabs, 5,990,225 snow crabs, 3,071,678 Tanner crabs, and 45,700 chinook salmon.

Total PSC limits for 2021 GOA groundfish trawl fisheries were 1,706 t of halibut mortality (and 257 t for longline fisheries) and 33,340 Chinook salmon. The PSC limits in the BSAI and GOA are further allocated among different fishing sectors, and in some cases, further apportioned among target fisheries, fishing seasons, or area. Attaining any of these limits closes the specific target fishery within the entire management area (for halibut and salmon PSC limits) or a specified closure area (for crab and herring PSC limits), for the remainder of the PSC period, season, or year, even if the target species catch limit has not been fully harvested.

The halibut and groundfish fisheries are also managed to allow the industry to find solutions to minimize

bycatch, typically to stay within regulated bycatch limits. Numerous exempted fishing permits (permits issued by NMFS to allow fishing activities that would otherwise be prohibited under federal regulations) have been issued to provide an opportunity for fishermen to test selective fishing techniques and gears, including halibut excluders and salmon bycatch reduction devices in trawl nets, deck sorting of halibut bycatch on trawl vessels to reduce discard mortality, the use of EM for estimating catch and bycatch on vessels using longline and pot gear, the use of EM for compliance monitoring on trawl vessels, and test the effectiveness of a voluntary program for vessels to avoid high bycatch rate areas that change within the fishing season as a way to reduce salmon bycatch in trawl fisheries.

Fisheries operating under catch share programs have significantly more opportunity to implement industry solutions across an entire fishery sector than do fisheries that operate under a race for fish system where individual vessels compete to catch as much fish as possible before the catch limit is attained. Implementation of an IFQ system for halibut and sablefish fixed gear fisheries, and the formation of cooperatives (a type of catch share program) in the Bering Sea pollock and bottom trawl fisheries and the Central Gulf of Alaska rockfish fisheries, have resulted in reduced bycatch and waste (NPFMC, 2014b, 2017). The race for fish was eliminated, which then allowed for development of more selective fishing gears and fishing practices, development of additional markets for lower valued species, and significant increases in utilization rates of fish landed (Fina, 2011; Strong and Criddle, 2013).

The BSAI catcher/processor bottom trawl fleet has the potential for their catch to be constrained by the halibut PSC limits established for this sector, so this sector has strived to develop solutions to minimize the bycatch and bycatch mortality of halibut. In 2015, the Council reduced the halibut PSC limit for this sector by 25% (to 1,745 t of halibut mortality) and requested that the sector's cooperative prepare annual halibut avoidance plans. Since then, representatives from the cooperative have annually reported back to the Council on the performance of the plans, and the agreement to ensure accountability of all participants in avoiding halibut. The cooperative's current halibut agreement includes three components: implementation of best practices for avoidance on the grounds (monitoring bycatch, use of excluders, communication protocols, and captains sharing information on locations to avoid), a halibut avoidance plan that defines standards and incentives to achieve good performance, and the use of deck sorting, which by regulation allows vessels to deck sort halibut and return halibut to the water quickly, thereby reducing halibut mortality and the mortality rate applied to the bycatch (ASC, 2021). In December 2021, the Council approved a measure that links the halibut PSC limits for this sector to indices of halibut abundance. At current abundance levels, this action will further reduce this sector's halibut PSC limits by 25% (to 1,309 t of halibut mortality).

The three BSAI pollock fleet cooperatives (inshore, at-sea, and mothership cooperative) operate under regulated bycatch limits for Chinook salmon which close the fishery if reached, and these cooperatives have established incentive plan agreements which require additional actions for individual operators to effectively avoid salmon on the fishing grounds at all levels of salmon and pollock abundance (Stram and Ianelli, 2014). These incentive plan agreements typically include requirements for salmon bycatch data reporting and information sharing; identification of bycatch hotspot areas where fishing by vessels with poor bycatch performance is prohibited; several areas with historically higher bycatch rates closed to all vessels; monetary penalties for vessels with consistently higher Chinook salmon bycatch rates relative to the fleet; in some cases incentives to avoid salmon by allowing a limited number of salmon credits (i.e., uncaught portion of a salmon cap) generated by using less than a vessels annual allocation of Chinook salmon bycatch units to carry over and thus providing an insurance buffer for future years; and a requirement that all pollock trawl nets contain salmon excluder devices to allow salmon to escape from the trawl net during fishing operations (Gruver, 2022; Madsen and Haflinger, 2022).

To ensure that BSAI pollock harvesters will make every effort to avoid salmon during fishing operations, the Council developed a salmon bycatch management program that is a blend of caps and incentives. The Chinook salmon caps are "hard" caps that if reached, close the entire pollock fishery for the year. In the Bering Sea, the overall bycatch limit is either 60,000 Chinook salmon or 45,000 Chinook salmon, depending on whether the previous year's Chinook run size was above or below an abundance threshold of 250,000 salmon (in aggregate) returns to the Kuskokwim River, Unalakleet River, and Upper Yukon River system (NMFS, 2016).

A key incentive to avoid salmon is the regulation whereby if the fleet exceeds the lower performance standard cap in 3 out of 7 years, it becomes the new hard cap in perpetuity. To avoid this possibility, the fleet voluntarily operates under lower performance standard caps of 47,591 salmon, or 33,318 salmon in years below the Chinook salmon abundance threshold, as a protection against reaching the overall hard cap, which would stop pollock fishing for the remainder of the year (Gruver, 2022). Representatives from the pollock cooperatives annually report to Council on the effectiveness of their efforts to avoid salmon in the previous year, and their incentive plans for the coming year.

Technological solutions have also been implemented to reduce bycatch and unobserved mortality. Biodegradable panels made of cotton thread are required for pot gear to minimize bycatch mortality associated with lost gear. Gillnets for groundfish fishing have been prohibited to prevent ghost

fishing and bycatch of non-target species. The use of bottom trawl gear for pollock fishing in the Bering Sea was prohibited to reduce crab, halibut, and other non-target bycatch. Research is currently underway to test the effectiveness of crab bycatch reduction in a variety of Pacific cod pot gear modifications, with the goal to develop a design that reduces crab bycatch while still catching target species, and then to encourage the voluntary use of those designs (BSFRF, 2022). The recent development of lightweight collapsible, "slinky pots," for sablefish, and the concomitant change in regulations to allow the use of pot gear to catch sablefish and halibut (instead of longline gear) has already reduced bycatch and further reductions are anticipated.

Regulatory approaches have also been used to limit discarding of groundfish species. Although all discards of groundfish count against the total catch quotas for each groundfish stock, discard of targeted species is considered especially wasteful as this is food that could be marketed and consumed. To address these concerns, regulations at 50 CFR 679.27 require full retention of all pollock, Pacific cod, and GOA shallow-water flatfish complex species (primarily *Lepidopsetta* sp. and *Pleuronectes* sp.), unless unfit for human consumption (i.e., decomposing fish that had previously been caught and discarded) or otherwise required to be discarded by regulation (e.g., fish caught in excess of MRA amounts). Full retention of rockfish (even if closed to directed fishing) in groundfish or halibut fisheries by catcher vessels using longlines, jigs, or pot gear is required to stop the waste of dead fish, as well as ensure accurate data collection and species identification. All sablefish caught in the IFQ fishery are required to be retained, provided there is sufficient quota share available on the vessel to cover the catch. However, due to the substantially reduced market value for smaller sablefish, participants in the sablefish fishery have requested the Council reevaluate the full retention requirement and allow fishermen to discard the smaller fish.

Methods

We evaluated the amount and composition of bycatch in the halibut and groundfish fisheries for the years 2013–21, based on the MSA definition of bycatch, using data generated by the NOAA Catch Accounting System (Cahalan et al., 2014). The Catch Accounting System uses data from observers and EM systems along with landings information to generate estimates of total groundfish catch, including at-sea discards, as well as estimates of PSC and other non-groundfish bycatch. Although bycatch data have been collected since 1991, only the data since 2013 are included in our evaluation of fishery specific bycatch amount to maintain consistency between estimation methods and data collection that have changed throughout time. Prior to 2013, for example, halibut bycatch was not reported for halibut and sablefish IFQ fisheries because there was no observer data from this fishery. Changes in estimation methodology did not impact PSC estimates relative to the PSC limits established for each year, therefore we summarized a longer time-series (1991–2021) of available PSC data to evaluate the bycatch of these species relative to PSC limits. PSC data are publicly available on the NMFS website.

We evaluated bycatch data for major fisheries in the North Pacific, with the fisheries defined based on the management area and gear used. These fishery groupings differ from those reported in the first edition of the National Bycatch Report (Karp et al., 2011), and, to a lesser extent, the updates of the report (Benaka et al., 2019). The Alaska fisheries in the National Bycatch Report updates (Benaka et al., 2019) are based on species targets (defined by the predominant species retained on a trip), management area, and gear used, but also, in some cases, the vessel type (catcher vessel or catcher/processor). The methodology results in splitting the data into specific fisheries that may not operate in a substantially different manner. For this paper, we lumped these fisheries together.

For example, in our analysis of the BSAI flatfish trawl fishery we did not distinguish between the Amendment 80 catcher/processor bottom trawl sector and the BSAI trawl limited access sector in the amount of bycatch and composition of discards, because both sectors use the same gear and fish in the same general area of the Bering Sea when fishing for yellowfin sole and other flatfish. Similarly, we examined bycatch in the Pacific cod longline fishery by area, without reporting bycatch separately for catcher vessels and catcher/processors. We also did not make a distinction based on whether or not a vessel used pelagic trawl or bottom trawl gear to harvest rockfish, as the biggest differences in fishery operations (including the rockfish species targeted) are between management areas.

For our fishery specific bycatch analysis, we treated bycatch data for Pacific halibut the same as bycatch data for groundfish and non-target species. Although halibut bycatch is typically reported in terms of bycatch mortality for PSC management, this mortality estimation is unique to halibut, therefore we did not adjust the bycatch data to account for mortality of discarded halibut in our analysis of fishery specific bycatch. This enabled an evaluation of bycatch and bycatch rates of halibut and all other species using a consistent methodology. However, in our analysis of PSC use relative to PSC limits for halibut, we did maintain the adjustment for discard mortality that is used for management purposes. Similarly, to maintain consistency with other estimations of total bycatch by fishery in this paper, we converted the bycatch data for salmon and crab from numbers of animals to total weight. Salmon and crab bycatch is typically monitored and managed by the number of animals caught, as is reported in our analysis of PSC use relative to limits. However, for the analysis of total discard weight, we applied the average observed weight of salmon

Figure 2.—Retained and discarded catch (t) in the groundfish and halibut fisheries in the Bering Sea and Aleutian Islands (BSAI) management area, by gear type.

or crab, by species, taken as bycatch in the groundfish fisheries to the total number of animals estimated to be taken as bycatch, to estimate total bycatch weight for each of these species.

Results

Total catch and discard by target gear type in the BSAI and GOA from 2013 through 2021 are shown in Figures 2 and 3. In both areas, pelagic trawl and pot fisheries had the lowest discard rates. Pelagic trawl fisheries had an average (2013–2021) discard rate of 1.1% in both the BSAI and GOA. Pelagic trawls are used to target walleye pollock, and to a lesser extent, some rockfish species in the GOA. Pot fisheries had an average discard rate of 3.4% in the BSAI and 5.0% in the GOA. There was a reduction in total catch in GOA pot fisheries starting in 2018 that was a result of reduced stock size and a corresponding closure of the directed fishery for Pacific cod. Bottom trawl fisheries had slightly higher discard rates, averaging 8.5% in the BSAI and 15.7% in the GOA. Total catch from bottom trawls has declined over the time series in both the BSAI and GOA due to reduced catches of flatfish and Pacific cod. Longline fisheries had the highest bycatch rates, averaging 18.8% in the BSAI and 46.4% in the GOA. A reduction in total catch by longline gear was due primarily to the reductions in Pacific cod catch limits over the time period. Reduced catches of sablefish in longline gear have also occurred as an increasing number of vessels have switched over to using pot gear to catch sablefish to reduce whale depredation.

The total amount of discards by species or species complex, across all groundfish fisheries (including jig gear) and the halibut fishery in the BSAI and GOA in 2021, is shown in Tables 1 and 2. In the BSAI, walleye pollock had the lowest discard rate (1.0%), but due to the high total catch of this species, it was the largest component (18.7%) of total discards across all fisheries. Discard rates for Atka mackerel, Pacific cod, Pacific ocean perch, and most flatfish species were under 10%. Higher discard rates

Figure 3. —Retained and discarded catch (t) in the groundfish and halibut fisheries in the Gulf of Alaska (GOA) management area, by gear type.

(30%–55%) were observed for other species, including sablefish, skates, and some rockfish species. Species that were fully accounted for and then discarded also include PSC species where retention is prohibited (salmon, crabs, Pacific herring) and unmarketable species (jellyfish, sculpins, sea stars, etc.). In the GOA, Pacific halibut was the largest component (23.4%) of the discards. Discard rates for pollock, sablefish, Pacific cod, and most rockfish were under 10%. Most flatfish had intermediate discard rates (14%–20%), and most skates were discarded (84%–97%). As with the BSAI, PSC species (salmon, crabs, Pacific herring) and unmarketable species (grenadier, sculpins, sea stars, etc.) were fully discarded in the GOA.

The 2021 catch and discard amounts, and discard amounts of the top ten species or species groups within the BSAI and GOA trawl fisheries are shown in Tables 3 and 4. Not surprisingly, the pelagic trawl fisheries primarily discarded pelagic species such as Scyphozoa jellies (primarily the northern sea nettle, *Chrysaora melanaster)*, squid (e.g., magistrate armhook squid, *Berryteuthis magister*), Pacific herring, smelt and other Osmerids (including capelin, *Mallotus villosus*, and eulachon, *Thaleichthys pacificus*), Pacific salmon, and walleye pollock. The bottom trawl fisheries tended to discard benthic species such as skates, flatfish, walleye pollock, and sculpins (including yellow Irish lord, *Hemilepidotus jordani*, in the GOA, and great sculpin, *Myoxocephalus polyacanthocephalus*, and plain sculpin, *M. jaok*, in the BSAI). Overall, walleye pollock and Scyphozoa jellyfish were the most discarded species in the BSAI trawl fisheries. Walleye pollock, arrowtooth flounder, *Atheresthes stomias*, and Pacific cod were the most discarded species in the GOA trawl fisheries. The largest total discard occurred in the BSAI flatfish fisheries (32,690 t), accounting for 34% of the total discards in the North Pacific halibut and groundfish fisheries. The BSAI pollock fishery had the second highest discard amount (19,761 t) of the trawl fisheries. Overall, the pollock fisheries had the lowest discard rates of all of the trawl fisheries, with 1.4% of the catch discarded in

Table 1.—Total discards (t) and retained catch (t) by species or species group, across all federally managed commercial halibut and groundfish fisheries in the Bering Sea and Aleutian Islands area, 2021.

Common Name	Scientific Name	Discard (t)	Retained (t)	Total (t)	% Discarded
Walleye pollock	*Gadus chalcogrammus*	14,161	1,363,936	1,378,097	1.0
Atka mackerel	*Pleurogrammus monopterygius*	706	60,647	61,354	1.2
Yellowfin sole	*Limanda aspera*	2,003	106,785	108,788	1.8
Pacific cod	*Gadus macrophalus*	3,615	132,100	135,715	2.7
Greenland turbot	*Reinhardtius hippoglossoides*	47	1,549	1,596	3.0
Kamchatka flounder	*Atheresthes evermanni*	223	6,444	6,667	3.3
Pacific ocean perch	*Sebastes alutus*	1,791	33,688	35,479	5.0
Rock soles	*Lepidopsetta polyxstra; L. bilineatus*	963	13,431	14,393	6.7
Flathead sole; Bering flounder	*Hippoglossoides elassodon; H. robustus*	774	9,485	10,259	7.5
Alaska plaice	*Pleuronectes quadrituberculatus*	1,282	14,579	15,862	8.1
Northern rockfish	*Sebastes polyspinus*	564	5,649	6,212	9.1
Arrowtooth flounder	*Atheresthes stomias*	984	8,029	9,014	10.9
Shortraker rockfish	*Sebastes borealis*	154	342	496	31.1
Other rockfish	*Sebastolobus alascanus; Sebastes* sp.	366	635	1,002	36.6
Sablefish	*Anoplopoma fimbria*	2,241	3,507	5,748	39.0
Rougheye/Blackspotted rockfish	*Sebastes aleutianus; S. melanostictus*	205	310	515	39.9
Skates	*Rajidae*	11,127	8,902	20,029	55.6
Other flatfish	*Pleuronectidae; Paralichthyidae*	1,708	930	2,638	64.8
Pacific halibut	*Hippoglossus stenolepis*	3,733	1,625	5,357	69.7
Octopus	*Octopoda*	148	22	170	87.3
Sharks	*Elasmobranchii*	352	7	359	98.0
Jellies	*Scyphozoa*	8,614	0	8,614	100.0
Sculpins	*Scorpaeniformes*	5,628	0	5,628	100.0
Squids	*Teuthoidea*	4,124	0	4,124	100.0
Giant grenadier	*Albatrossia pectoralis*	2,677	0	2,677	100.0
Sea stars	*Asteroidea*	2,401	0	2,401	100.0
Pacific herring	*Clupea pallasii*	1,878	0	1,878	100.0
Non-Chinook salmon	*Oncorhynchus keta;* other *O.* sp.	1,222	0	1,222	100.0
Misc. fish sp.	*Teleostomi*	315	0	315	100.0
Red king crab	*Paralithodes camtschaticus*	309	0	309	100.0
Tunicates	*Tunicata*	220	0	220	100.0
Sponge unidentified	*Porifera*	170	0	170	100.0
Tanner crab	*Chionoecetes bairdi*	151	0	151	100.0
Sea anemones	*Actinaria*	142	0	142	100.0
Snails	*Gastropoda*	126	0	126	100.0
Snow crab	*Chionoecetes opilio*	84	0	84	100.0
Eelpouts	*Zoarcidae*	55	0	55	100.0
Invertebrates (unident.)	*Invertebrata*	50	0	50	100.0
Chinook salmon	*Oncorhynchus tshawytscha*	49	0	49	100.0
Golden king crab	*Lithodes aequispina*	44	0	44	100.0
Corals/Bryozoans	*Bryozoa*	28	0	28	100.0
Urchins/dollars/cucumbers	*Echinodermata*	19	0	19	100.0
Misc. crabs (unident.)	*Decapoda*	19	0	19	100.0
Brittle stars	*Ophiurodea*	15	0	15	100.0
Grenadiers (unident.)	*Macrouridae*	13	0	13	100.0
Bivalves	*Bivalvia*	10	0	10	100.0
Sea pens/sea whips	*Octocoralia*	8.7	0.0	8.7	100.0
Hermit crabs	*Paguroidae*	8.3	0.0	8.3	100.0
Black/blue/dark rockfish	*Sebastes melanops; S. ciliatus; S. mystinus*	5.7	0.0	5.7	100.0
Pandalid shrimp	*Pandalus* sp.	4.7	0.0	4.7	100.0
Greenlings	*Hexagammidae*	3.1	0.0	3.1	100.0
Misc. crustaceans (unident.)	*Crustaceamorpha*	1.7	0.0	1.7	100.0
Other osmerids	*Osmeridae*	1.4	0.0	1.4	100.0
Lanternfishes	*Myctophidae*	0.7	0.0	0.7	100.0
Saffron cod	*Eleginus gracilis*	0.6	0.0	0.6	100.0
Misc. invertebrates	*Invertebrata*	0.5	0.0	0.5	100.0
Polychaete worms	*Polychaeta*	0.4	0.0	0.4	100.0
Pacific sandfish	*Trichodon trichodon*	0.3	0.0	0.3	100.0
Pricklebacks	*Stichaeidae*	0.2	0.0	0.2	100.0
Pacific sand lance	*Ammodytes hexapterus*	0.2	0.0	0.2	100.0
Misc. deepwater fish	*Teleostomi*	0.1	0.0	0.1	100.0
Smelts	*Osmeridae*	0.1	0.0	0.1	100.0
Gunnels	*Pholidae*	0.0	0.0	0.0	100.0
Total BSAI		75,553	1,772,602	1,848,155	4.1

Table 2.—Total discards (t) and retained catch (t) by species or species group, across all federally managed commercial halibut and groundfish fisheries in the Gulf of Alaska area, 2021.

Species Group Name	Scientific Name	Discard (t)	Retained (t)	Total (t)	% Discarded
Walleye pollock	*Gadus chalcogrammus*	1,347	99,814	101,160	1.3
Northern rockfish	*Sebastes polyspinus*	38	2,339	2,376	1.6
Pacific ocean perch	*Sebastes alutus*	517	28,383	28,900	1.8
Dusky rockfish	*Sebastes variabilis*	106	2,822	2,929	3.6
Sablefish	*Anoplopoma fimbria*	1,206	15,295	16,501	7.3
Pacific cod	*Gadus macrophalus*	1,407	17,769	19,176	7.3
Thornyhead rockfish	*Sebastolobus* sp.	23	284	306	7.4
Demersal shelf rockfish	*Sebastes ruberrimus; other Sebastes* sp.	14	122	136	10.3
Atka mackerel	*Pleurogrammus monopterygius*	107	832	939	11.4
Rex sole	*Glyptocephalus zachirus*	44	257	301	14.5
Rougheye/blackspotted rockfish	*Sebastes aleutianus; S. melanostictus*	79	342	421	18.7
Arrowtooth flounder	*Atheresthes stomias*	1,885	8,144	10,029	18.8
Shallow water flatfish	*Pleuronectidae*	379	1,478	1,858	20.4
Flathead sole	*Hippoglossoides elassodon*	145	562	708	20.5
Shortraker rockfish	*Sebastes borealis*	198	372	571	34.7
Pacific halibut	*Hippoglossus stenolepis*	5,041	9,235	14,276	35.3
Other rockfish	*Sebastes* sp.	487	753	1,240	39.2
Octopus	*Octopoda*	29	27	55	51.7
Deep water flatfish	*Pleuronectidae*	79	17	96	82.4
Big skate	*Beringraja binoculata*	664	122	786	84.5
Longnose skate	*Raja rhina*	1,198	105	1,303	92.0
Skates	*Rajidae*	859	27	886	96.9
Sharks	*Elasmobranchii*	2,091	24	2,114	98.9
Giant grenadier	*Albatrossia pectoralis*	931	0	931	100.0
Sculpins	*Scorpaeniformes*	903	0	903	100.0
Misc. fish sp.	*Teleostomi*	634	0	634	100.0
Squids	*Teuthoidea*	276	0	276	100.0
Smelts	*Osmeridae*	241	0	241	100.0
Grenadier (unident.)	*Macrouridae*	236	0	236	100.0
Sea stars	*Asteroidea*	94	0	94	100.0
Other osmerids	*Osmeridae*	89	0	89	100.0
Chinook salmon	*Oncorhynchus tshawytscha*	53	0	53	100.0
Black/blue/dark rockfish	*Sebastes* sp.	30	0	30	100.0
Pacific herring	*Clupea pallasii*	19	0	19	100.0
Tanner crab	*Chionoecetes bairdi*	19	0	19	100.0
Jellyfish	*Scyphozoa*	15	0	15	100.0
Corals/bryozoans	*Bryozoa*	8.7	0.0	8.7	100.0
Non-Chinook salmon	*Oncorhynchus* sp.	8.3	0.0	8.3	100.0
Snails	*Gastropoda*	6.8	0.0	6.8	100.0
Sea anemones	*Actinaria*	6.3	0.0	6.3	100.0
Greenlings	*Hexagammidae*	4.6	0.0	4.6	100.0
Sponge unidentified	*Porifera*	2.7	0.0	2.7	100.0
Misc. crabs (unident.)	*Decapoda*	2.5	0.0	2.5	100.0
Pricklebacks	*Stichaeidae*	2.0	0.0	2.0	100.0
Urchins/dollars/cucumbers	*Echinodermata*	2.0	0.0	2.0	100.0
Saffron cod	*Eleginus gracilis*	1.8	0.0	1.8	100.0
Eelpouts	*Zoarcidae*	0.9	0.0	0.9	100.0
Golden king crab	*Lithodes aequispina*	0.7	0.0	0.7	100.0
Invertebrates (unident.)	*Invertebrata*	0.4	0.0	0.4	100.0
Pandalid shrimp	*Pandalus* sp.	0.4	0.0	0.4	100.0
Sea pens/sea whips	*Octocoralia*	0.4	0.0	0.4	100.0
Brittle stars	*Ophiurodea*	0.2	0.0	0.2	100.0
Hermit crabs	*Paguroidae*	0.1	0.0	0.1	100.0
Bivalves	*Bivalvia*	0.1	0.0	0.1	100.0
Pacific sand lance	*Ammodytes hexapterus*	0.1	0.0	0.1	100.0
Red king crab	*Paralithodes camtschaticus*	0.1	0.0	0.1	100.0
Tunicates	*Tunicata*	0.1	0.0	0.1	100.0
Pacific sandfish	*Trichodon trichodon*	0.1	0.0	0.1	100.0
Pacific hake	*Merluccius productus*	0.1	0.0	0.1	100.0
Misc. crustaceans (unident.)	*Crustaceamorpha*	0.1	0.0	0.1	100.0
Lanternfishes	*Myctophidae*	0.0	0.0	0.0	100.0
Gunnels	*Pholidae*	0.0	0.0	0.0	100.0
Misc. invertebrates	*Invertebrata*	0.0	0.0	0.0	100.0
Misc. deepwater fish	*Teleostomi*	0.0	0.0	0.0	100.0
Snow crab	*Chionoecetes opilio*	0.0	0.0	0.0	100.0
Total GOA		21,531	189,126	210,658	10.2

Table 3.—Amount (t) and composition of discards in the 2021 BSAI trawl fisheries.

Pollock trawl		Flatfish trawl		Rockfish/mackerel trawl		Pacific cod trawl	
Scyphozoa jellies	7,817	Pollock	11,944	Pacific o. perch	819	Pollock	289
Squid sp.	3,822	Skate sp.	3,037	Atka mackerel	541	Pacific halibut	125
Pacific herring	1,708	Sculpin sp.	2,453	Northern rockfish	451	Skate sp.	102
Pacific cod	1,477	Sea star sp.	2,151	Pollock	440	Other flatfish sp.	89
Chum salmon	1,212	Giant grenadier	1,977	Sculpin sp.	425	Pacific cod	80
Pollock	968	Pacific halibut	1,862	Giant grenadier	407	Rock sole	63
Sablefish	662	Other flatfish sp.	1,374	Skate sp.	348	Atka mackerel	57
Pacific o. perch	652	Alaska plaice	1,281	Sablefish	309	Sculpin sp.	51
Skate sp.	341	Yellowfin sole	1,249	Other rockfish	252	Scyphozoa jellies	51
Shark sp.	300	Sablefish	874	Misc. fish sp.	174	Arrowtooth fl..	44
Other species	802	Other species	4,490	Other species	1,167	Other species	92
Total discard	19,761	Total discard	32,690	Total discard	5,335	Total discard	1,043
Total catch	1,368,493	Total catch	227,565	Total catch	109,891	Total catch	20,303
% Discard	1.4	% Discard	14.4	% Discard	4.9	% Discard	5.1

Table 4.—Amount (t) and composition of discards in the 2021 GOA trawl fisheries.

Pollock trawl		Flatfish trawl		Rockfish trawl		Pacific cod trawl	
Pollock	583	Arrowtooth fl.	700	Arrowtooth fl.	648	Pollock	249
Squid sp.	226	Pacific cod	596	Giant grenadier	388	Arrowtooth fl.	122
Smelt sp.	218	S.W. flatfish sp.	320	Other rockfish sp.	345	Giant grenadier	78
Other osmerid sp.	89	Pacific halibut	308	Pacific cod	309	Pacific halibut	43
Shark sp.	63	Sablefish	285	Pollock	248	Big skate	22
Grenadier sp.	46	Pacific o. perch	253	Pacific o. perch	242	Sculpin sp.	20
Pacific halibut	40	Pollock	241	Pacific halibut	199	S.W. flatfish sp.	17
Chinook salmon	34	Sculpin sp.	152	Sablefish	175	Chinook salmon	12
Misc. fish sp.	22	Flathead sole	85	Misc. fish sp.	169	Pacific cod	11
Pacific o. perch	21	Misc. fish sp.	81	Atka mackerel	89	Misc. fish sp.	10
Other species	43	Other species	342	Other species	408	Other species	28
Total discard	1,386	Total discard	3,361	Total discard	3,220	Total discard	612
Total catch	114,099	Total catch	16,133	Total catch	28,142	Total catch	2,417
% Discard	1.2	% Discard	20.8	% Discard	11.4	% Discard	25.3

the BSAI, and 1.2% in the GOA. The highest bycatch rates among the trawl fisheries were those targeting flatfish, with 14.4% of the catch discarded in the BSAI, and 20.8% in the GOA, and for the trawl fishery targeting Pacific cod in the GOA (25.3%).

Total catch and discards from BSAI and GOA longline and pot fisheries, and the discard amounts of the top ten species or species groups in these fisheries, are shown in Tables 5 and 6. The highest amounts of total discard in the fixed gear fisheries were in the BSAI Pacific cod longline fishery (12,767 t) and the GOA halibut longline fishery (8,711 t). The BSAI and GOA Pacific cod and sablefish pot fisheries had very low amounts of discard, totaling 2,426 t, and low discard rates, ranging from 2.6% to 6.1%. Sculpins and yellowfin sole in the BSAI, and sculpins and Pacific halibut in the GOA comprised the highest weight of bycatch in the Pacific cod pot fisheries. The majority of bycatch in the sablefish pot fisheries consisted of arrowtooth flounder, sablefish, Pacific halibut, and giant grenadier, *Albatrossia pectoralis*. The highest bycatch rates for fixed gear were in the halibut and sablefish longline fisheries, with 55.0%–63.3% of the catch discarded in the BSAI, and 28.5%–48.2% in the GOA. Giant grenadier, shark species (including Pacific spiny dogfish, *Squalus suckleyi*, and Pacific sleeper shark, *Somniosus pacificus*) and sablefish comprise the highest bycatch amount in the sablefish longline fisheries. Pacific halibut, sharks, skates, and Pacific cod comprised the largest bycatch by weight in the halibut longline fishery. The Pacific cod longline fisheries had more intermediate discard rates (15.7%–27.0%) with skates, Pacific halibut, and sculpins accounting for much of the bycatch.

Table 7 shows the total catch and discards from all 2021 halibut and groundfish fisheries the BSAI and GOA. Of the 2,058,806 t of fish (including PSC and non-target species) caught in the BSAI and GOA halibut and groundfish fisheries, 97,083 t (4.7%) were discarded as bycatch. Overall, discard rates were low in the trawl fisheries (3.6%) and pot fisheries (4.4%), and higher in the longline fisheries (23.5%). The rates for trawl and pot fisheries were similar in the BSAI and GOA. Discard rates in the GOA longline fisheries (40.0%) were higher than the BSAI longline fisheries (17.7%), due to differences in scale of the Pacific cod and Pacific halibut fisheries in each area. No bycatch rate estimates were made for other gears (jig and halibut pots) as bycatch information is incomplete for these small fisheries. Data from the jig fishery include catch and discards of groundfish, but do not include PSC or bycatch of other non-target species, which are not estimated for the jig fishery.

Table 5.—Amount (t) and composition of discards in the 2021 BSAI longline and pot fisheries.

Pacific cod longline		Pacific cod pot		Sablefish longline		Sablefish pot		Halibut longline	
Skate sp.	6,557	Sculpin sp.	404	Giant grenadier	226	Arrowtooth fl.	34	Skate sp.	726
Sculpin sp.	2,138	Yellowfin sole	370	Skate sp.	16	Giant grenadier	24	Pacific cod	587
Pacific halibut	1,086	Red king crab	221	Pacific halibut	11	Golden king crab	16	Pacific halibut	333
Pacific cod	897	Octopus sp.	117	Sculpin sp.	1	Pacific halibut	10	Sablefish	236
Pollock	482	Sea star sp.	92	Arrowtooth fl.	1	Sablefish	6	Sculpin sp.	113
Yellowfin sole	358	Snail sp.	51	Sablefish	1	Pacific cod	5	Pollock	34
Flathead sole	241	Sablefish	43	Rougheye rockf.	1	Kamchatka fl.	3	Other rockfish sp.	19
Arrowtooth fl.	203	Pacific cod	36	Other flatfish sp.	1	Misc. crab sp.	2	Sea star sp.	17
Other flatfish sp.	118	Pacific halibut	25	Sea anemone sp.	1	Greenland turbot	2	Giant grenadier	13
Sea star sp.	112	Snow crab	23	Kamchatka fl.	0	Shark sp.	0	Arrowtooth fl.	8
Other species	576	Other species	97	Other species	2	Other species	3	Other species	25
Total discard	12,767	Total discard	1,479	Total discard	260	Total discard	106	Total discard	2,112
Total catch	81,296	Total catch	34,346	Total catch	411	Total catch	1,982	Total catch	3,840
% Discard	15.7	% Discard	4.3	% Discard	63.3	% Discard	5.3	% Discard	55.0

Table 6.—Amount (t) and composition of discards in the 2021 GOA longline and pot fisheries.

Pacific cod longline		Pacific cod pot		Sablefish longline		Sablefish pot		Halibut longline	
Pacific halibut	527	Sculpin sp.	55	Shark sp.	499	Arrowtooth fl.	246	Pacific halibut	3,673
Skate sp.	280	Pacific halibut	45	Giant grenadier	387	Sablefish	186	Shark sp.	1,266
Shark sp.	168	Pacific cod	25	Sablefish	236	Pacific halibut	75	Longnose skate	932
Big skate	141	Tanner crab	15	Grenadier sp.	170	Giant grenadier	64	Sculpin sp.	602
Longnose skate	88	Sea star sp.	15	Longnose skate	134	Rougheye rockf.	11	Skate sp.	473
Sablefish	55	Octopus sp.	13	Pacific halibut	128	Shark sp.	10	Big skate	449
Sculpin sp.	47	Sablefish	9	Shortraker rockf.	112	Grenadier sp.	9	Pacific cod	408
Pacific cod	46	Misc. fish sp.	9	Skate sp.	68	D. W. flatfish sp.	8	Misc. fish sp.	302
Pollock	24	S.W. flatfish sp.	4	Rougheye rockf.	52	S.W. flatfish sp.	6	Sablefish	260
Arrowtooth fl.	24	Other rockfish sp.	3	Arrowtooth fl.	44	Pacific cod	6	Arrowtooth fl.	98
Other species	53	Other species	8	Other species	113	Other species	17	Other species	248
Total discard	1,452	Total discard	204	Total discard	1,943	Total discard	637	Total discard	8,711
Total catch	5,385	Total catch	7,852	Total catch	6,809	Total catch	10,482	Total catch	18,057
% Discard	27.0	% Discard	2.6	% Discard	28.5	% Discard	6.1	% Discard	48.2

Table 7.—Total catch (t) and discard (t) in the 2021 GOA and BSAI halibut and groundfish fisheries.

Area	Gear	Catch (t)	Discard (t)	% Discard
BSAI	Trawl	1,726,252	58,829	3.4%
	Longline	85,547	15,139	17.7%
	Pot	36,328	1,585	4.4%
	Other gears	31	0	n/a
	Total BSAI	1,848,158	75,553	4.1%
GOA	Trawl	160,791	8,579	5.3%
	Longline	30,251	12,106	40.0%
	Pot	18,344	841	4.6%
	Other gears	1,272	4	n/a
	Total GOA	210,658	21,530	10.2%
Total	Trawl	1,887,043	67,408	3.6%
	Longline	115,818	27,245	23.5%
	Pot	54,662	2,426	4.4%
	Other gears	1,303	4	n/a
	Total all areas	2,058,816	97,083	4.7%

Figures 4–6 show the time series of PSC relative to overall PSC limits. Overall, the bycatch of crabs in the BSAI and Pacific halibut in both the BSAI and GOA have been substantially reduced since the early 1990's. In recent years in BSAI fisheries, bycatch of Pacific herring and chum salmon was higher than average, whereas Chinook salmon bycatch was relatively lower. Bycatch of all PSC species has been well below total PSC limits. There was only one recent instance where a PSC amount was exceeded (BSAI Pacific herring PSC in 2020). Chinook salmon bycatch in the GOA has been variable without a clear trend. Note that the increased PSC limit for Chinook salmon in the GOA occurred when new PSC limits were implemented for the non-pollock trawl fisheries in 2015, in addition to the Chinook PSC limits previously implemented for the pollock fisheries.

Discussion

Our results indicate that management of bycatch in the North Pacific halibut and groundfish fisheries has been effective at minimizing bycatch to the extent practicable, as measured by the low bycatch amounts relative to total catch in recent years. In 2021 for example, only 4.7% (97,083 t) of the catch was discarded as bycatch, which was similar to the 2017–2021 average of 4.8% (106,291 t/year). This relatively low bycatch and bycatch rate reflects the rationalization of many fisheries through implementation of catch share systems, as well as changes in regulations, capital investment, ex-

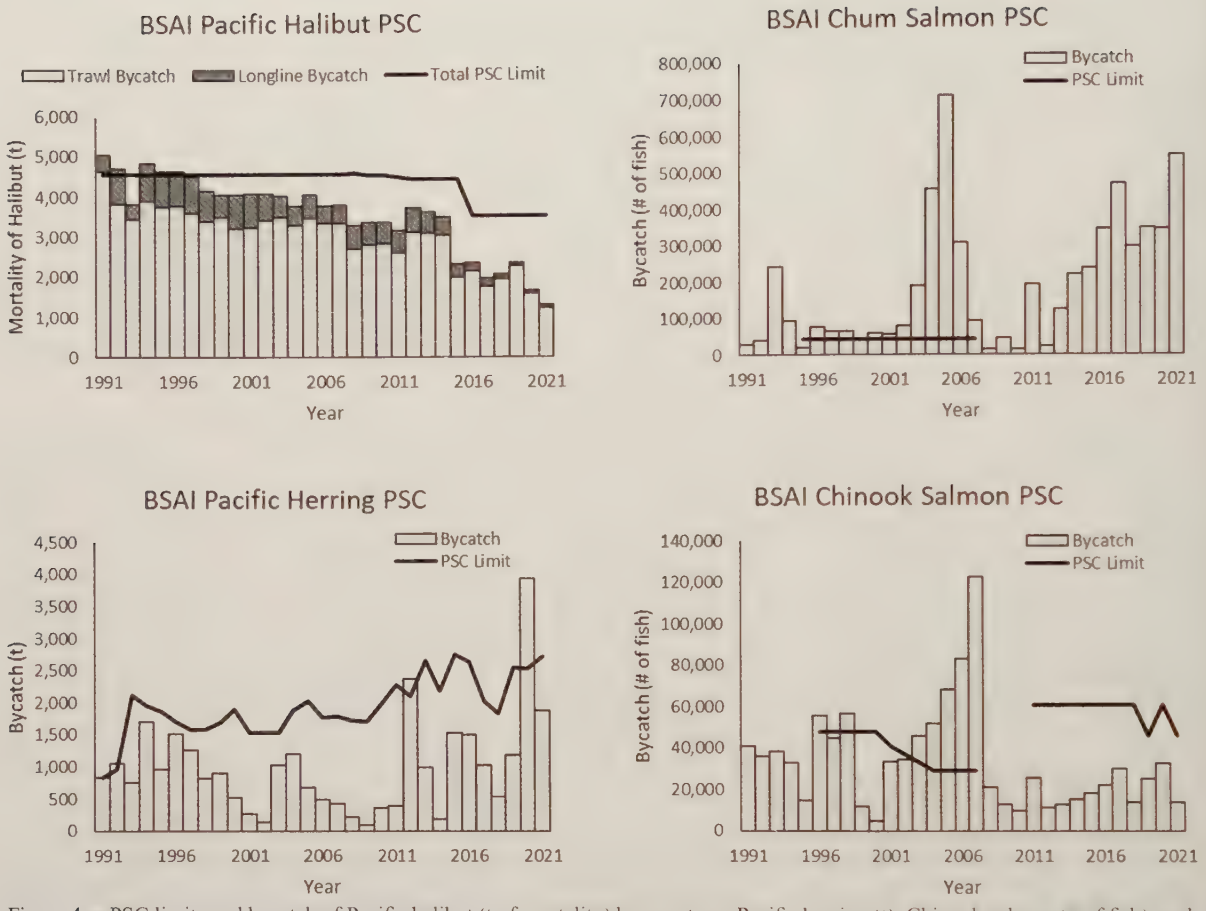

Figure 4.—PSC limits and bycatch of Pacific halibut (t of mortality) by gear type, Pacific herring (t), Chinook salmon (# of fish), and chum salmon in BSAI groundfish fisheries, 1991–2021.

panded markets, and other factors that have changed since the 1990's. Bycatch in BSAI groundfish fisheries during the period 1990–1994 ranged from 198,000 t to 315,000 t annually, with corresponding bycatch rates of 12% to 16%. In the GOA, bycatch ranged from 41,000 t to 61,000 t annually with bycatch rates of 17% to 21% during this time period (Queirolo et al., 1995). By the 1998–2000 period, total bycatch in the BSAI and GOA groundfish fisheries had been reduced to 149,000 t to 151,000 t (8.4%–9.8% of the total catch) (Smoker, 2010). Our results indicate that further reductions in total bycatch and bycatch rates have occurred over the past 20 years.

The current bycatch rate for the North Pacific halibut and groundfish fisheries is among the lowest in the world. Recent global bycatch estimates range from 4.1%–4.5% in the Southeast Pacific and Eastern Indian Ocean to 23%–29% in the western Central Atlantic and Southwest Atlantic (Pérez Roda et al., 2019). Overall, about 10% of the global catch is discarded based on recent estimates (Zeller et al., 2017; Pérez Roda et al., 2019; Gilman et al., 2020). Global bycatch estimates from the 1990's and early 2000's had estimated global bycatch at 35% to 40% of the catch (Alverson et al., 1996; Davies et al., 2009).

Our results show relatively low bycatch rates occur across most North Pacific fisheries gear types. Average global discard rates (including North Pacific bycatch data) were 12% for pelagic otter trawls, 31% for bottom otter trawls, and 24% for bottom longlines (Pérez Roda et al., 2019). For comparison, the 2021 North Pacific discard rates were 1% for pelagic otter trawls (BSAI and GOA pollock trawl fisheries), 11% for bottom trawls (sum of BSAI and GOA flatfish, rockfish, and Pacific cod trawl fisheries), 24% for bottom longlines (sum of BSAI and GOA Pacific cod, sablefish, and halibut longline fisheries), and 4% for pot gear (sum of BSAI and GOA Pacific cod and sablefish pot fisheries). A global comparison with pot gear was not made because global discard rates for pot gear include fisheries for crus-

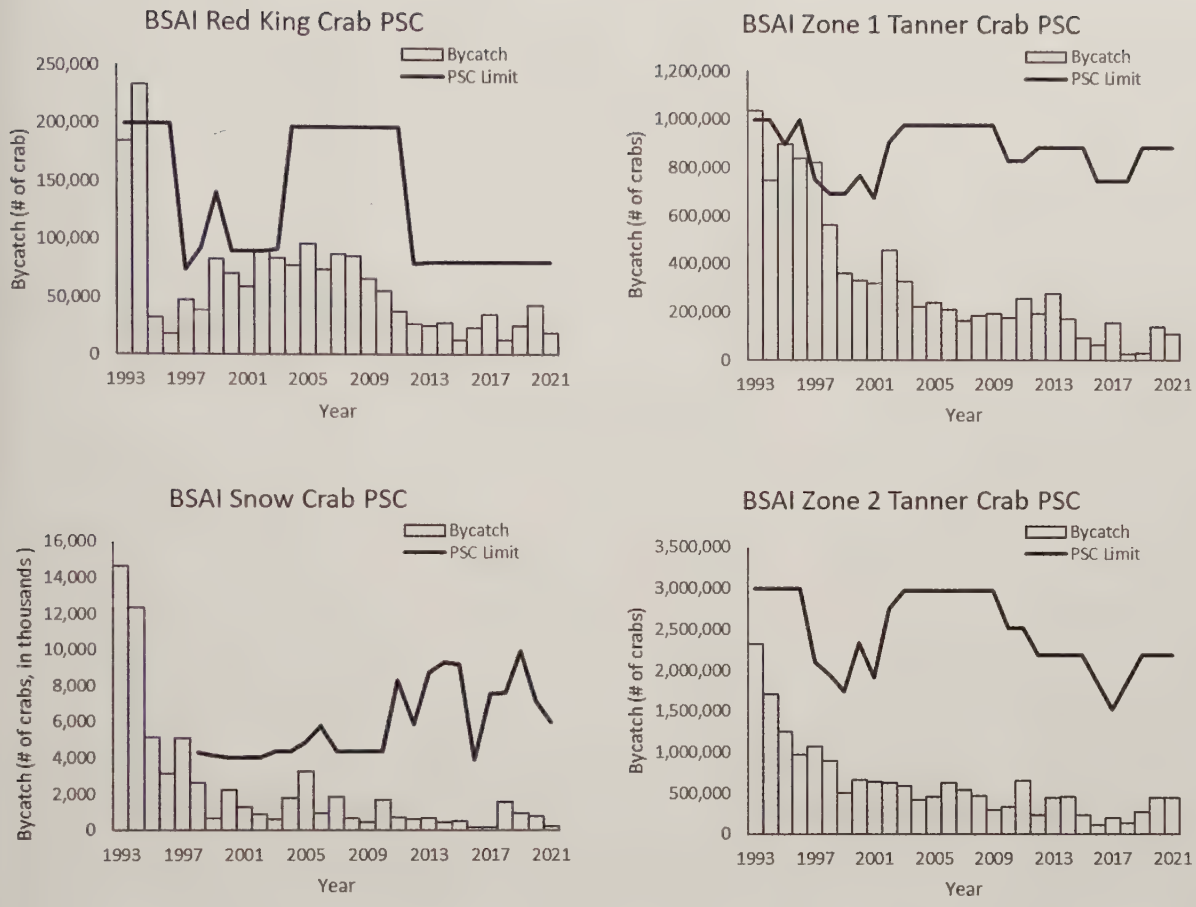

Figure 5.—PSC limits (# of crab) and bycatch (# of crab) of red king crab, snow crab, and Zone 1 (Bering Sea statistical areas 508, 509, 512, and 516) and Zone 2 (Bering Sea statistical areas 513, 517, and 521) Tanner crab in BSAI groundfish fisheries, 1993 2021.

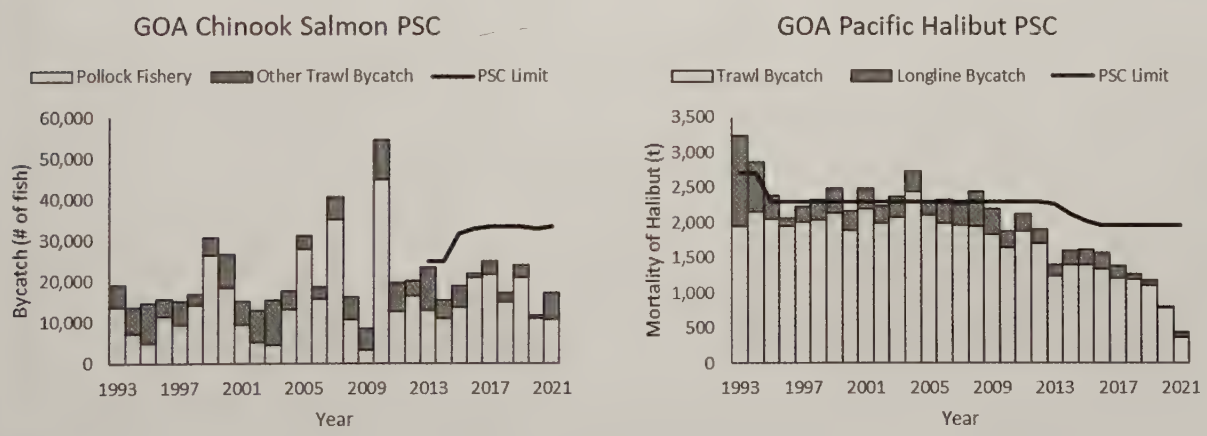

Figure 6.—PSC limits and bycatch of Chinook salmon (# of fish) by trawl target fishery and Pacific halibut (t of mortality) by gear type in GOA groundfish fisheries, 1993–2021.

taceans, which have higher discard rates than demersal fish (Gilman et al., 2020).

The total amount of bycatch, the composition of the bycatch, and the bycatch rate differ greatly among the fisheries and gear type used in those fisheries (Tables 3–6), and this information can be used to focus management efforts on minimizing total bycatch or minimizing bycatch rates. The 2021 Bering Sea pollock trawl fishery had a very low bycatch rate (1.4% discarded), but because it is a high-volume fishery, had a high amount of bycatch (19,761 t), consisting mostly of Scyphozoa jellies (40%), followed by squid (19%) and herring (9%) (Table 3). Although these jellyfish are not marketable, squid can be retained and processed into food or bait, and management could provide additional incentives to retain squid and other species. On the other end of the range, the sablefish longline fisheries and the halibut fishery off Alaska (BSAI and GOA combined) had relatively high bycatch rates (30.5% and 49.4%, respectively) but since these fisheries were lower in volume, had lower total bycatch amounts (2,203 t and 10,823 t, respectively). The rapid conversion of the sablefish fishery from longlines to pots to avoid depredation by sperm whales, *Physeter macrocephalus*, in the GOA and orcas, *Orcinus orca*, in the BSAI, has likely reduced bycatch rates and total bycatch in the sablefish fishery after pot gear was authorized in 2017, based on our data. Our results indicate that the use of pot gear in the sablefish fisheries reduces bycatch of sharks and skates in particular compared to longline gear. Recent authorization for the use of pot gear to target Pacific halibut (also to avoid whale depredation) could result in similar reductions in bycatch should an effective and acceptable pot gear be developed for this species.

Total bycatch of PSC species has been well below the established PSC limits in recent years. The PSC limits for Chinook salmon and Pacific halibut can be very restrictive for trawl fisheries in the GOA and BSAI, and the fleet expends substantial effort and added cost to minimize bycatch, avoid attaining the limits and thus avoid shutting down the fishery (Stram and Ianelli, 2014). As a result, Pacific halibut bycatch in the GOA and BSAI groundfish fisheries is currently about one-third of what it was in the early 1990's (Fig. 4, 6). The original PSC limits for BSAI Chinook salmon (implemented in 1996) and chum salmon (implemented in 1995) were seasonal triggers which closed specified fixed areas despite where salmon were located. These salmon PSC limits failed to constrain total salmon bycatch in the Bering Sea, so they were replaced with caps on Chinook salmon PSC that if reached would close the pollock fishery, a system of inseason closure areas dependent on real-time PSC rates, and other avoidance requirements by industry contract, which proved more successful at minimizing Chinook salmon bycatch.

Bycatch amounts of crab PSC in the BSAI trawl fisheries are a small fraction of what they were in the 1990's due to a number of regulatory and non-regulatory changes, including more modernized trawl fishing gear, a requirement that bottom trawls must use elevated sweeps (Rose et al., 2012), establishment of static closure areas to protect crab (Witherell and Woodby, 2005; Kruse et al., 2010), changes in the abundance of crab stocks, reduced PSC limits, and implementation of catch share cooperatives for trawl fisheries which serve to slow down fishing effort and allow for sector-wide efforts to minimize bycatch. Crab industry representatives have recently requested the Council further reduce PSC limits for crab and take other actions to rebuild depleted BSAI crab stocks. Because crab and other PSC species are mobile and move in response to environmental or other variables, further reduction in PSC for bottom trawl fisheries could potentially be attained with expanded use of dynamic temporary closures based on real-time data, similar to how the BSAI pollock fleet avoids areas of high Chinook salmon bycatch (Pons et al., 2021).

In the North Pacific, conservation concerns regarding bycatch are limited. All Pacific halibut and groundfish caught, whether taken in a target fishery or as bycatch in another fishery, accrue towards the annual catch limits and are also factored into the stock assessment and specification of annual catch limits (Witherell et al., 2000; Stewart and Webster, 2020; Kong et al., 2021). From a biological perspective, so long as bycatch is considered a subset of catch, it makes no difference to the fish population whether the catch is targeted or caught incidentally, or whether the catch is retained or discarded (Tagart, 2004). Additionally, all discards in the groundfish fishery are assumed dead for accounting purposes, even though some proportion of fish and invertebrates discarded survive capture and release from commercial fishing gears (Benoit, 2013; Conners and Levine, 2016) with survival rates depending on environmental and other factors (Davis, 2002). Bycatch of non-target species, which are not managed with annual catch limits but which have the potential to be impacted by the halibut and groundfish fisheries, is also monitored to the extent data are available. Some non-target species commonly taken as bycatch such as grenadier, sculpins, and forage fish, are assessed regularly to ensure that stocks are not being overfished (Reuter et al., 2010; Rodgveller and Siwicke, 2020, Ormseth and Yasumiishi, 2021). Other ecosystem components, including bycatch species, are monitored to the extent data are available and reported annually in ecosystem status reports (Ferriss and Zador, 2021; Ortiz and Zador, 2021; Sidden, 2021). Clearly, bycatch can be a major conservation issue in regions where bycatch is uncontrolled, unquantified, unassessed, or not included in the total removals prior to setting an annual catch limit (Harrington et al., 2005; Komoroske and Lewison, 2015), but that is not the case for federally managed fisheries in the North Pacific.

There are social and conservation concerns with the bycatch of Chinook salmon and chum salmon off Alaska. Scientific data indicate that Chi-

nook salmon and chum salmon population trends in this region are driven by environmental factors such as marine heatwaves and the prey quantity and quality that occur in the first few months after outmigration of smolts (Howard et al., 2019; Murphy et al., 2021). Nevertheless, salmon bycatch has been a particularly challenging issue for the Council given the economic and cultural importance and subsistence value of these species to Alaska Natives and people living in remote areas of the State of Alaska.

Social concerns are heightened when subsistence salmon fisheries managed by the State of Alaska are limited or prohibited. For example, closures of subsistence salmon fishing in the Yukon River and Kuskokwim River regions in coastal western Alaska in 2021 and 2022 has put an intense focus on the bycatch of salmon in groundfish fisheries. In 2021, a total of 15,827 Chinook salmon and 535,282 chum salmon were taken as bycatch in Bering Sea groundfish fisheries, most of which were in the pollock fishery. Because Chinook salmon and chum salmon from other countries in the Pacific region intermingle in the ocean and are caught as bycatch, NMFS annually conducts extensive and statistically reliable genetic sampling of salmon taken as bycatch in the pollock trawl fisheries so that the impacts of bycatch on spawning stocks in different regions can be estimated (Ianelli and Stram, 2014; Faunce, 2015). Genetic analysis of salmon taken in the 2020 Bering Sea pollock fishery showed that about 52% of the Chinook salmon bycatch was estimated to have originated from coastal western Alaska and about 9% of the chum salmon caught as bycatch originated from coastal western Alaska (Guthrie et al., 2022; Barry et al., 2022). Overall, since 2011, Chinook salmon bycatch has reduced the runs by an average of 1.9% for the aggregate coastal western Alaska stocks and 0.6% for the Upper Yukon River stock (NMFS et al., 2022). Chum salmon bycatch is roughly estimated to have affected aggregate western Alaska stocks by about 1.1%; the average impact rate (2004 through 2011) for coastal western Alaska was 0.46% and 1.16% impact rate on the Upper Yukon fall chum salmon run (NPFMC, 2012). Although these impacts appear to be small, conservation concerns about western Alaska salmon stocks and lack of subsistence harvest opportunities have prompted the Council to continue evaluating ways to further minimize bycatch of salmon consistent with the MSA.

Regulatory discards are difficult to address but could be a focus for future management action. In the North Pacific, regulatory discards typically result from prohibitions on retaining PSC, discarding any amount of fish of a species closed for directed fishing that exceeds the MRA specified for that species in that fishery, or mandatory discarding of a target species if the total allowable catch (TAC) limit is reached to eliminate incentives to catch the species. Development of an alternative tool to MRA's, such as implementation of procedures by cooperatives, could reduce the need for MRA's in these catch share fisheries. Further reductions in regulatory discards could result with implementation of fully rationalized trawl fisheries in the GOA whereby all TAC is allocated and can be redistributed among participants during the season. While there are no regulatory discards of groundfish due to size limits, the minimum size limit for Pacific halibut creates regulatory discards in the halibut IFQ fishery, which results in halibut being the prominent discarded species in that fishery in the GOA (Table 6). Eliminating the size limit could reduce bycatch, but this action may have impacts on long term yield of the halibut fishery (Martell et al., 2015; Stewart et al., 2020).

Regulatory discards could be reduced by authorizing the retention of PSC species, particularly for those fish that are dead or likely to die if discarded. Current regulations do authorize some voluntary retention of salmon and halibut taken as bycatch, but only for donation to hunger relief organizations. Although not all the halibut and salmon PSC are retained for donation, the non-profit organization SeaShare has worked with the Alaska seafood industry to distribute 2,660 t of donated salmon and halibut to foodbanks throughout the U.S. (Watson et al., 2020). Discards could be further reduced if regulations allowed the retention of all salmon, Pacific halibut, and Pacific herring, in the groundfish fisheries. While potentially challenging from a policy perspective, allowing for the sale of these species would provide some incentive to retain and process the incidental catch. Increasing retention of bycaught crabs; however, would be more difficult for most vessels participating in the groundfish fisheries, in that vessels would need circulating seawater tanks onboard to keep the crabs alive until processing.

Economic discards can be further reduced or even eliminated by development of new regulations. Several different management tools have been used to minimize economic discards in North Pacific groundfish fisheries. Economic discards have been reduced by establishing catch share programs that allocate the TAC—minus the amount allocated to other fisheries or needed to cover incidental catch in other fisheries (the incidental catch allowance) where applicable—to individual permit holders as IFQ (sablefish and halibut fisheries), to community development quota entities, or to cooperatives (the BSAI pollock cooperatives, bottom trawl catcher/processor cooperatives, GOA rockfish program cooperatives, and a pending program for trawl catcher vessels targeting Pacific cod). These catch share-based programs eliminated the race for fish, allowing operators to be more selective in the size and species of fish being targeted and retained, resulting in lower bycatch (Fina, 2011; Grimm et al., 2012; NPFMC, 2014b, 2017). For example, implementation of the cooperative catch share program for the BSAI bottom trawl catcher/processor fleet reduced discards from about 22% to less than 10% (NPFMC, 2014b). Implementation of catch share programs in unrationalized fisheries

could further reduce discarding. Another way to eliminate economic discards is to simply require that all fish are retained. This has been the goal for the European Union as an approach to eliminate waste of edible protein (Karp et al., 2019). In the North Pacific, 50 CFR 679.27 regulations require that all walleye pollock, Pacific cod, and GOA shallow water flatfish caught be retained unless otherwise regulated (e.g., retention only up to MRA allowance, or if the species is on PSC status) or dead or decomposing fish previously caught and discarded and thus unfit for human consumption. Although bycatch is just one component of the waste of edible seafood in the U.S., waste reduction is needed for food security with a growing global population living under the stress of climate change (Love et al., 2015). In the North Pacific, the waste of fish from bycatch (97,083 t in 2021) is considerably less than the amount of groundfish quota that goes unharvested (422,791 t in 2021) due to PSC limits and other regulatory impediments, and economic factors, as well as the amount of walleye pollock that remains unharvested in some years due to the 2 million t OY limit in the BSAI.

In taking any action to minimize bycatch, the Council must make policy decisions relative to balancing trade-offs between National Standard 1 (achieve Optimum Yield) and other National Standards, including National Standard 9 (minimize bycatch to the extent practicable). To date, these decisions balancing National Standards have considered the ability of the fleet to catch the TAC and achieve OY while minimizing bycatch to the extent practicable, taking into account the biological and/or socio-economic impacts of any bycatch. The Council must decide at what level bycatch has been practically minimized to meet the requirements of the MSA.

Recent legislation introduced in the 117th Congress proposed to revise MSA National Standard 9 by eliminating the words "to the extent practicable" (Huffman, 2021). The word "practicable" includes consideration of social and economic tradeoffs in deciding how much additional regulatory burden and cost the fishery should bear to minimize bycatch or further reduce the amount of bycatch in the fishery while achieving OY. Without the phrase "to the extent practicable," the Council could be under pressure to minimize bycatch regardless of the net costs to the nation in terms of food security, economic activity, fishing community health, or jobs. Eliminating the phrase "to the extent practicable" could also increase litigation risk for actions taken to minimize bycatch, given it is impossible to completely eliminate bycatch unless all catch is required to be retained (Tagart, 2004).

Climate change is affecting stock size, composition, productivity, and distribution of groundfish in the North Pacific (Mueter et al., 2021; Hollowed et al., 2022), which will impact the amount and composition of bycatch in future years. Some fish populations (e.g., pollock, Pacific cod) and crab populations are expanding into the northern Bering Sea and the high Arctic in response to warmer water on the Bering Sea shelf and loss of sea ice and a reduced cold pool. These environmental conditions are further projected to substantially reduce the pollock and Pacific cod stocks over the long term (Holsman et al., 2020) and are having significant effects on current crab stocks. Environmental monitoring and modeling revealed that marine heatwaves in the GOA during 2014–16 drastically reduced the forage base available and increased metabolism of Pacific cod, resulting in a drastic reduction in stock size and recruitment (Barbeaux et al., 2020). These same conditions appear to have been favorable for sablefish, with historically large year-classes of sablefish produced during 2016–18 period (Goethel et al., 2021). The abundance of these large year-classes has resulted in substantially higher trawl discards of sablefish, which raises management and allocation issues. Additional marine heatwaves appear likely to be more common, and future changes in ocean conditions in the North Pacific will impact the magnitude and species composition of catch and discards in ways that are difficult to predict. Fishery managers must develop climate resilient fishery management plans and regulations using ecosystem-based approaches to take into account this uncertainty. As noted by Holsman et al. (2020), the current ecosystem-based management approach used in the North Pacific can ameliorate climate change impacts on fisheries in the near-term, but the long-term benefits of this approach are limited by the magnitude of anticipated change. Revised management approaches, including those taken to minimize bycatch to the extent practicable, will be required.

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Building Capacity for Kelp (Laminariales) Mariculture in the Salish Sea: Ecological Considerations and Initial Farm Design Guidance

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Introduction

Kelp (Phylum Phaeophyta, Order Laminariales) are large brown seaweeds found in coastal temperate and polar waters around the world. As photosynthesizing organisms, brown seaweeds grow in the photic zone and create widespread marine "forests" that provide crucial habitat to nearshore fishes, invertebrates, and mammals (Steneck et al., 2002; Klinger, 2015). Kelp has a biphasic life history—the large sporophyte plants obligately alternate with a microscopic gametophyte (sexual) phase. Kelp forests are aptly named, as the sporophytes of different kelp species assume a diversity of structural roles; some grow to form a sea surface canopy, while others lie prostrate along the seafloor or grow into the water column to form an "understory" (Wernberg and Filbee-Dexter, 2019).

Although many kelps are found in these subtidal, diverse, three-dimensional structures, they can also proliferate in small patches outside of kelp forests in areas such as the rocky intertidal zone. Sporophytes of different kelp species can live anywhere from a single growing season to several decades and demonstrate variations in successional establishment, making the habitat "hotspots" they form temporally dynamic and variable (Dayton, 1985; Reed et al., 2006; Springer et al., 2010).

In addition to providing habitat and refugia, kelp provide the marine environment with energy and nutrients via large amounts of organic material (whole plants, detritus, and dissolved organic material) that serve as the base of nearshore, deepwater, and terrestrial food webs (Steneck et al., 2002; Christie et al., 2009; von Biela et al., 2016; Elliott Smith and Fox, 2022). They also influence the water quality surrounding them, changing carbon chemistry, nitrogen, and other nutrients, metals, and chemicals (Klinger, 2015; Pfister et al., 2019). These ecosystem services, defined as the direct or indirect benefit to human society from the natural environment (Kumar, 2010), can be assigned a monetary value and may be considered in conservation or management decision-making.

Kelp mariculture has increased worldwide and the global scientific literature increasingly highlights local ecological benefits. This has captured the interest of many aspiring Washington kelp farmers who are eager to harness the ecological and economic benefits of cultivating kelp, even though the regulatory process for permitting kelp farming is still a work in progress. This paper is a response to the growing local interest and regulatory uncertainty in a kelp farming industry, and it was specifically developed to provide potential farmers, leasees of state-owned aquatic lands, and regulators with pertinent biological and ecological information that may assist those writing and assessing permit and lease applications. This paper was also developed for tribes that are considering seaweed farming and/or may review proposed farms for environmental effects or impacts to Usual and Accustomed fishing areas. Permitting for seaweed farms and similar activities are particularly complex in Washington state, where proposals require multiple local, state, and federal reviews as well as tribal reviews (Conroy et al., 2023). Here we consider the

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ABSTRACT—As a new type of marine activity in an environmentally and socially complex region, one limitation of the development of seaweed farming within Washington State waters of the Salish Sea is a lack of information about how to best implement sustainable farming practices. In this paper we discuss the potential environmental benefits and risks posed by open-water native kelp (Laminariales) mariculture in the Salish Sea, suggest initial farm design guidance to maximize ecosystem benefits and minimize ecological harms, and identify priority areas for future monitoring and research. We also emphasize the importance of conducting kelp farming in a way that accommodates tribal concerns and ensures access to treaty-protected fishing grounds. We suggest an adaptive management approach, using the best available information to propose initial guidance followed by suggestions for data collection that would inform the efficacy of those considerations. Our goal is to provide regulatory and proprietary agencies, tribes, and potential sea farmers with the best available, locally-specific (where possible) information, making suggestions when appropriate, and identifying remaining knowledge gaps to inform future research.

global literature on the ecological impacts of kelp farming, paired with the regional ecological and social context in which local kelp mariculture could occur, and highlight the precautions and challenges potential farmers and marine resource managers should consider. We pay particular attention to any potential negative ecological impacts and marine spatial use conflicts, especially including treaty rights and related concerns from western Washington treaty tribes.[1] We are taking this approach in recognition of the social and ecological controversy associated with other forms of sea farming in the region (e.g., intensive fish farming), which has proven detrimental to their social acceptance, and which the emerging kelp farming in the U.S. portion of the Salish Sea should strive to avoid.

We provide farm design guidance for open-water native kelp cultivation, using an approach that considers two main dimensions: 1) potential ecological impacts associated with kelp cultivation, specifically within a regional context, and 2) the different aquatic stages of the kelp farming cycle, including site selection, cultivation, and harvest (the ex situ hatchery phase and post-harvest processes are not discussed). The relative newness of domestic kelp cultivation, especially on the U.S. west coast, presents both limitations and opportunities. Data relevant to local kelp species (emphasized in Hollarsmith et al., 2022, also see Table 1 and Table 2), environmental conditions, and impacts are generally lacking. Therefore, we have tried to fill information gaps by referring to published literature from other areas, relying on insights from local researchers, and emphasizing the need for adaptive management informed by future research and monitoring. As such, the majority of information provided pertains to sugar kelp, *Saccharina latissima*; however, the initial farm design guidance and ecological considerations presented are also applicable to other types of seaweed mariculture (e.g. other kelps, see Table 1), dulse, *Palmaria mollis* (Druehl and Clarkston, 2016); nori, *Pyropia/Porphyra* sp. (Bergdahl, 1990). Although we recognize that kelp cultivation can serve various purposes, including restoration, bioremediation, and mitigation, we focus here on commercial kelp cultivation. As the opportunities for kelp cultivation in the Salish Sea increase, these other important uses should be considered in more depth—either independently or in tandem with commercial cultivation (i.e. "restorative aquaculture" (Mizuta et al., 2022)).

Regional Description

Ecological Context

The Salish Sea is the marine area of the southern portion of British Columbia, Canada, and the northwestern portion of Washington. The major waterways that define it include the Strait of Georgia, the Strait of Juan de Fuca, and Puget Sound (Fig. 1). The Puget Sound portion of the Salish Sea has several interconnected basins or biogeographic regions that encompass contiguous, ecologically unique, and spatially isolated freshwater, estuarine, and marine habitats (Downing, 1983; Burns, 1985).

These natural conditions have contributed to the genetic differentiation of many species. A few examples, among better known native species that have been studied, include several rockfish species—yelloweye rockfish, *Sebastes ruberrimus*; copper rockfish, *Sebastes caurinus*; brown rockfish, *Sebastes auriculatus*; quillback rockfish, *Sebastes maliger*—and a roundfish, Pacific hake, *Merluccius productus,* within the various regions of the Salish Sea and between Puget Sound and the open ocean (Seeb, 1998; Buonaccorsi et al., 2002, 2005; Iwamoto et al., 2015; Andrews et al., 2018). Genetic differentiation has also been found among bull kelp, *Nereocystis luetkeana*, within the Strait of Juan de Fuca, the San Juan Islands, and central and south Puget Sound (Gierke, 2019), but genetic analysis has not occurred for other kelp species in the study area. Evidence for limited genetic differentiation has been demonstrated in several native molluscan shellfish species within the Salish Sea, but generally more so between Puget Sound and locations along the outer coast, including the Olympia oyster, *Ostrea lurida* (Heare et al., 2017; Silliman, 2019), geoduck, *Panopea generosa* (Vadopalas et al., 2004, 2012); and basket cockle, *Clinocardium nuttali* (Dimond et al., 2022).

The region is also home to several species listed under the Endangered Species Act (ESA), including distinct populations of Chinook, *Oncorhynchus tshawytscha*, and chum, *Oncorhynchus keta*, salmon; bull trout, *Salvelinus confluentus*; green sturgeon, *Acipenser medirostris*; eulachon, *Thaleichthys pacificus*; yelloweye rockfish; and bocaccio, *Sebastes paucispinis*; in addition to southern resident killer whales, *Orcinus orca*; humpback whales, *Megaptera novaeangliae*; and marbled murrelet, *Brachyramphus marmoratus*; with designated critical habitat for a subset of these species within marine waters of the study area. Threatened invertebrates in the southern Salish Sea include the native Olympia oyster, *Ostrea lurida*, and the pinto abalone, *Haliotis kamschatkana*, which has seen a 97% decline since 1992 (Carson and Ulrich, 2019).

The nearshore of the region is generally defined as encompassing the marine shoreline, intertidal beach, and subtidal zone to the photic zone depth (about 30 m/98 ft deep). It hosts a rich variety of aquatic vegetation including canopy forming bull kelp, *Nereocystis luetkeana*; (regional distribution shown in Fig. 1) and eelgrass, *Zostera* sp. Eelgrass is a prolific type of submerged nearshore vascular vegetation that provides important ecosystem services including substrate for the spawn of Pacific herring, *Clupea pallasi*, and habitat for myriad fish and invertebrate species (Plummer et al., 2013).

[1]Although the term "tribe" has largely fallen out of favor among academics, it is the term that local indigenous groups use in reference to themselves and their communities (e.g., Treaty Indian Tribes in Western Washington, 2011).

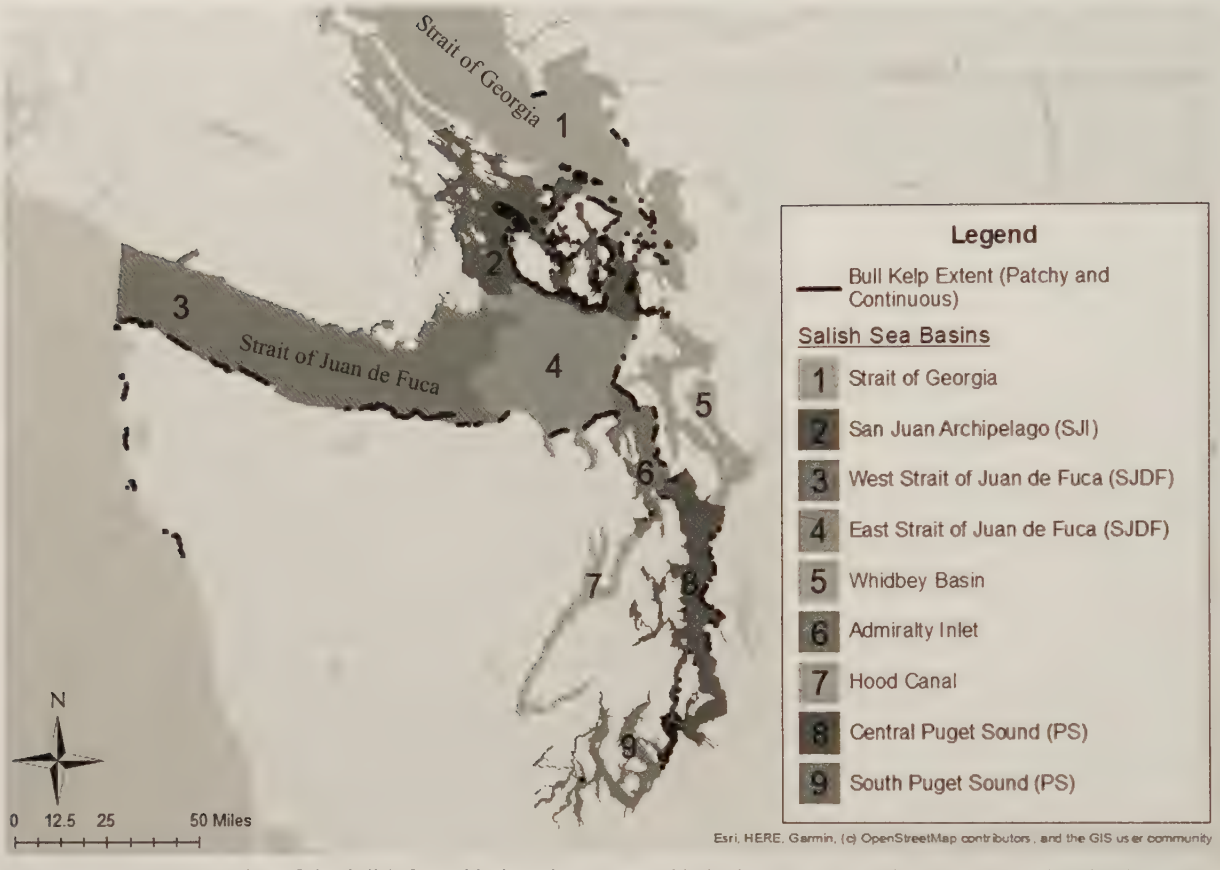

Figure 1.—The U.S portion of the Salish Sea with the sub-oceanographic basins (ERMA Northwest, 2022) and bull kelp extent (patchy and continuous) (WDNR, 2023).

Eelgrass meadows are generally found in shallower waters, between +1.4 m to -12.0 m relative to mean lower low water (MLLW, Hannam, 2013), then kelp beds. Seventeen species of native kelp reside in Washington waters of the Salish Sea (Calloway et al., 2020), and widespread bull kelp losses, especially in the central and southern basins have been documented (Berry et al., 2019, 2021). A Kelp Conservation and Recovery Plan for Puget Sound was completed in 2020 (Calloway et al., 2020), and bull kelp along the Pacific Coast were petitioned for ESA listing in 2022 (Kelkar and Carden, 2022).

About 40% of Puget Sound's nearshore area has been altered through dock and pier development, shoreline bulkheading, single family housing, and industrial port development (Fresh et al., 2011). As such, conservation and restoration of the nearshore environment has been a regional priority for decades (Kelkar and Carden, 2022).

Social and Legal Context

In addition to the complexity and uniqueness of the natural environment, the waters of the Salish Sea are among the commercially, and recreationally, busiest along the west coast. For instance, the Ports of Seattle and Tacoma constitute the fourth largest container gateway in North America (Community Attributes, 2019). In addition, several hundred thousand boat-based recreational fishing trips occur every year, typically in the summer months (WDFW, 2012). Washington has the largest commercial shellfish industry in the United States, and there are hundreds of thousands of recreational shellfish harvest trips annually (Washington Sea Grant, 2015). Competing uses of the sea surface and water column create potential conflict with other end users, making the emergence of a kelp farming sector potentially problematic.

Importantly, the study area is part of the ancestral territory of 16 western Washington treaty tribes with unique legal rights and political capacities, such as adjudicated Usual and Accustomed (U and A's)[2] fishing areas

[2]Usual and accustomed fishing areas (U and A's) are spatially delimited areas of marine space fished by tribal community ancestors. U and A's are tribe-specific and place-based but not exclu-

in marine waters (Singleton, 2009), as well as several other federally-recognized tribes and tribes without federal recognition. Following the "Fish Wars" of the 1960's and a contentious legal battle, the Boldt Decision (U.S. vs. Washington, 1974) reaffirmed the fishing rights reserved by Western Washington tribes in treaties negotiated with Governor Isaac Stevens in the mid-1850's on behalf of the U.S government. The tribes secured recognition as marine resource co-managers and rights to 50% of the harvest from their U and A's (NWIFC, 2014). Similarly, in 1994 a Federal court ruled that tribes have treaty rights to harvest shellfish in Puget Sound, including on private lands (U.S. vs. Washington, 1994) unless otherwise excluded by legal agreement with the Tribes.

Western Washington treaty tribes are understandably watchful of territorialized activities (e.g., mariculture, marine protected areas, etc.) that could compromise resource access or tenure rights within their U and A's (Singleton, 2009). They also advocate for habitat protections under treaty rights (ie. the "Culverts Case", U.S. vs. Washington, 2017), which kelp mariculture may support. The decline of finfish fisheries in the Salish Sea has likely contributed to tribal pursuit of additional income sources. Many tribes currently have shellfish mariculture operations and several are interested in adding kelp farming for commercial and/or subsistence use as well as for potential habitat enhancement benefits (Washington Sea Grant, 2023).

Mariculture in Washington has been increasingly subject to public controversy, including litigation in recent years. The use of pesticides in Willapa Bay and Grays Harbor to control burrowing shrimp, *Neotrypaea californiensis* and *Upogebia pugettensis*, native species that adversely affect oyster production, was subject to public scrutiny and litigation and the pesticide application permit was ultimately denied by Washington's Department of Ecology (ECY).[3] Similarly, existing net pen operations and shellfish permitting regimes have been litigated. In 2017, the spill from net pens of several hundred thousand farmed Atlantic salmon, *Salmo salar*, in north Puget Sound generated lawsuits and led to Washington's legislature phasing out the raising of non-native fish in the marine environment (Klein, 2021). In 2020 the U.S District Court for the Western District of Washington vacated the Army Corps of Engineers' (ACOE) Nationwide 48 Permit for shellfish activities in the state. The litigation was initiated by several non-profit organizations and the Swinomish Tribe, and it centered around concerns about intertidal and subtidal habitats of fish, eelgrass, *Zostera* sp., and birds, marine substrate, non-native and native species, pollution, and water quality (Klein, 2020). Similarly, in the 1980's two proposed Washington seaweed farms were stopped by concerns about impacts to shoreline view-scapes and fishing ground accessibility (Egan, 1988).

Although the sociopolitical complexities of marine space use in Puget Sound present a significant challenge to the development of commercial kelp cultivation, the human use of seaweed resources in the Salish Sea is likely as old as the human settlement of the region. Archeological evidence suggests that the Americas may have first been settled by maritime peoples following the rich assemblage of marine resources found in the kelp forests that extend along the Pacific Rim from Japan all the way down to Chile (Erlandson et al., 2007; Erlandson et al., 2015). Based on explorer travelogs, early ethnography, and tribal lore and contemporary practices, kelp has played an important role in traditional technology, subsistence activities, and stories, and continues to be an important part of Pacific Northwest indigenous traditional ecological knowledge (Naar, 2020). Currently, recreational and subsistence harvest of wild seaweeds is allowed in Washington via a Department of Fish and Wildlife (WDFW) seaweed and shellfish license (RCW 79.135.410), and commercial harvest of seaweed that fouls shellfish cultivation gear is allowed via permit by the Department of Natural Resources (DNR) and registration with WDFW through an Aquatic Farm Registration (RCW 15.85.020; WAC 220-370-060).

Potential Ecological Impacts

Our approach to developing farm design guidance for kelp cultivation begins with a consideration of the potential ecological impacts of growing kelp on longlines in open marine waters. The broader Pacific Northwest region, which also includes coasts of Oregon and British Columbia, is considered one of the 25 marine ecoregions worldwide with the greatest potential for restorative kelp farming that could mitigate nutrient pollution, habitat loss, ocean acidification, and trawl fishing pressure, as well as enhance socioeconomic and human health (Theuerkauf et al., 2019). Several recent reviews have discussed the possible ecological impacts associated with kelp farming, many of which emphasize the compelling evidence that kelp cultivation provides significant positive effects for its surrounding environment, including the following ecosystem services: habitat provision, food web support, water quality remediation, carbon sequestration, and shoreline buffering (Campbell et al., 2019; Grebe et al., 2019; Langton et al., 2019; Gentry et al., 2020; Barrett et al., 2022, Forbes et al., 2022). However, these reviews have also stressed the importance of continued monitoring, research, and mitigation measures applied in an adaptive management context to maximize potential benefits while minimizing potential harms.

Building on these broader reviews, we present here a list of possible ecological impacts in the specific context of the southern Salish Sea, flagging issues of particular concern and noting the availability of local data (Ta-

sive. Thus, the same area can include the U and A's of multiple tribes, and non-treaty resource users are also able to access and harvest from the area (Singleton, 2009).

[3]Doenges R. 2018. Letter to Ken Wiegardt, President of Willapa Grays Harbor Oyster Growers Association. 27 Sept. 2018.

Table 1.—Socio-ecological considerations for kelp species native to the Salish Sea.

	Alaria marginata	Costaria costata	Cymathaere triplicata	Hedophyllum sessile	Laminaria setchellii
Common name	Broad-winged kelp, Ribbon kelp	Five-ribbed kelp, Seersucker kelp	Three-ribbed kelp, Triple rib, Fold-rib kelp	Sea cabbage, Sweet kombu	Southern stiff-stiped kelp, Split blade kelp, Wild N. Pacific kombu
Washington distribution[1]	SJDF, SJI	PS, OC	SJDF, SJI, Whidbey Island (west side)	SJDF, SJI (west side)	Outer SJDF, OC on most exposed areas
Ocean parameters Temperature	Mortality, no recruitment >18 °C (O'Clair and Lindstrom, 2000; Muth et al., 2019)[2]	Mortality >20 °C, no recruitment >18°C (O'Clair and Lindstrom, 2000; Muth et al., 2019)[2]	Mortality 15-18 °C (O'Clair and Lindstrom, 2000)[2]	Mortality >15°C (O'Clair and Lindstrom, 2000)[2]	5-15°C optimal; reproduction Inhibited >17 °C; no recruitment >18 °C (O'Clair and Lindstrom, 2000; Bartsch et al., 2008; Muth et al., 2019)[2]
Nitrate	1-10 µmol/L; optimum 10 µmol/L (Muth et al., 2019)[3]	1-10 µmol/L; optimum 5 µmol/L at 12°C (Muth et al., 2019)[3]			1-10 µmol/L; optimum 10 µmol/L (Muth et al., 2019)[3]
Salinity/turbidity					
Substrate	Rocky (O'Clair and Lindstrom, 2000)	Rocky or woody (O'Clair and Lindstrom, 2000)	Rocky (O'Clair and Lindstrom, 2000)	Rocky (O'Clair and Lindstrom, 2000)	Rocky (O'Clair and Lindstrom, 2000)
Depth	Mid-low intertidal, 3 m (O'Clair and Lindstrom, 2000)	Low intertidal and upper subtidal, >3 m (O'Clair and Lindstrom, 2000)	Low intertidal and upper subtidal, >3 m (O'Clair and Lindstrom, 2000)	Mid to low intertidal, 3 m (O'Clair and Lindstrom, 2000)	Extreme low intertidal and upper subtidal, <10 m (O'Clair and Lindstrom, 2000)
Wave action	Fully sheltered to fully exposed (Mumford, 2007)	Fully sheltered to fully exposed (Mumford, 2007)	Moderately exposed (Mumford, 2007)	Moderate to fully exposed (O'Clair and Lindstrom, 2000)	Moderately sheltered to fully exposed (O'Clair and Lindstrom, 2000)
Life history Canopy form	Prostrate (Gabrielson et al., 2006; Mumford, 2007)	Prostrate (Gabrielson et al., 2006; Mumford, 2007)	Prostrate (Gabrielson et al., 2006; Mumford, 2007)	Prostrate (Gabrielson et al., 2006; Mumford, 2007)	Stipitate (Gabrielson et al., 2006; Mumford, 2007)
Life cycle (sporophyte phenology)	Annual (Luning, 1991)	Annual or perennial (O'Clair and Lindstrom, 2000)	Annual (Mondragon and Mondragon, 2010)	Perennial (2–3 yr.) (O'Clair and Lindstrom, 2000)	Perennial (up to 14 yr) (Germann, 1987)
Sporophyte size	<3 m (Luning, 1991)	2 m (O'Clair and Lindstrom, 2000)	4 m (Mondragon and Mondragon, 2010)	150 cm (O'Clair and Lindstrom, 2000)	1.5 m (O'Clair and Lindstrom, 2000)

Macrocystis pyrifera	Nereocystis luetkeana	Pleurophycus gardneri	Postelsia palmaeformis	Pterygophora californica	Saccharina latissima	Lessoniopsis littoralis
Giant kelp	Bull kelp, Bullwhip kelp	Broad-ribbed kelp, Sea spatula	Sea palm	Woody-stemmed kelp, Old growth kelp	Sugar kelp, Sugar wrack	Flat pompom kelp, Strap kelp, Ocean ribbons
SJDF (west of Low Point)	PS, OC	Straits, SJI, Whidbey Is (west side)		SJDF, SJI; rare in central and southern PS	PS, OC	OC, outer SJDF
<16.3 °C optimal; recruitment can occur at 18 °C (O'Clair and Lindstrom, 2000; Schiel and Foster, 2006; Muth et al., 2019) [2]	10-15 °C optimal; no recruitment >18 °C (Vadas, 1972; Maxell and Miller, 1996; Muth et al., 2019) [2]	14-15 °C triggers senescence; mortality >15 °C (O'Clair and Lindstrom, 2000; Germann, 2011; Pfister and Betcher, 2018) [2]	No recruitment >18 °C (Muth et al., 2019) [2]	No recruitment >18 °C (Muth et al., 2019) [2]	10-15 °C optimal; mortality >19 °C (O'Clair and Lindstrom, 2000; Simonson et al., 2015) [2]	No recruitment >18 °C (Muth et al., 2019) [2]
1-10 µmol/L; optimum 10 µmol/L at 12°C (Luning, 1991; Muth et al., 2019) [3]	1-10 µmol/L; optimum 10 µmol/L (Muth et al., 2019) [3]	Low NO_3^- leads to blade loss; nitrogen-limited (NO_3^-, NH_4^+) (O'Clair and Lindstrom, 2000; Germann, 2011) [3]	1-10 µmol/L; optimum 10 µmol/L (Muth et al., 2019) [3]	1-10 µmol/L (Muth et al., 2019) [3]	Saturation at 10-20 µmol/L (Simonson et al., 2015) [3]	1-10 µmol/L; optimum 5 µmol/L (Muth et al., 2019) [3]
Low salinity poorly tolerated at high temperatures (Mumford, 2007) [4]	Range 26-31 o/00, but 28-29 o/00 ideal; wide salinity tolerance if sedimentation is low (Maxell and Miller, 1996; Mumford, 2007; Lind, 2016) [4]				26-31 o/00; tolerates low salinity regardless of temperature (Mumford, 2007; Lind, 2016) [4]	
Rocky (O'Clair and Lindstrom, 2000)	Cobble and rocky (O'Clair and Lindstrom, 2000)	Rocky (O'Clair and Lindstrom, 2000)	Rocky (O'Clair and Lindstrom, 2000)	Cobble and rocky (Druehl and Clarkston, 2016)	Mixed; rock, shell, debris, and even sand in CA (O'Clair and Lindstrom, 2000)	Rocky, very exposed (O'Clair and Lindstrom, 2000)
Lower intertidal to deep subtidal, 5-30+ m (Schiel and Foster, 2006; Druehl and Clarkston, 2016)	Subtidal, 3-17 m (Mumford, 2007)	Extreme low intertidal and upper subtidal, <15 m (O'Clair and Lindstrom, 2000)	High to mid intertidal, <3 m (Luning, 1991)	Low intertidal to subtidal, 2-20 m (typically 10 m) (Dayton, 1985; Mumford, 2007)	Lower intertidal and upper subtidal, <30 m (O'Clair and Lindstrom, 2000)	Low intertidal, 3 m (O'Clair and Lindstrom, 2000)
Fully exposed; wave action with 2-4 cm/s current speed enhances N uptake (Schiel and Foster, 2006; Mumford, 2007)	Fully sheltered to fully exposed; withstands strong currents (O'Clair and Lindstrom, 2000; Mumford, 2007)	Strong currents (O'Clair and Lindstrom, 2000)	Fully exposed; adapted to extreme wave shock (O'Clair and Lindstrom, 2000; Mondragon and Mondragon, 2010)	Fully sheltered to moderately exposed, strong currents (Mumford, 2007)	Fully sheltered to fully exposed (O'Clair and Lindstrom, 2000)	Extremely exposed habitats, heavy surf (O'Clair and Lindstrom, 2000)
Floating (Gabrielson et al., 2006; Mumford, 2007)	Floating (Gabrielson et al., 2006; Mumford, 2007)	Prostrate (Gabrielson et al., 2006; Mumford, 2007)	Stipitate (Gabrielson et al., 2006; Mumford, 2007)	Stipitate (Gabrielson et al., 2006; Mumford, 2007)	Prostrate (Gabrielson et al., 2006; Mumford, 2007)	Stipitate (Gabrielson et al., 2006; Mumford, 2007)
Perennial (O'Clair and Lindstrom, 2000)	Annual (Dayton, 1985; O'Clair and Lindstrom, 2000)	Perennial, deciduous (3–6 yr) (O'Clair and Lindstrom, 2000; Pfister and Betcher, 2018)	Annual (O'Clair and Lindstrom, 2000)	Perennial; (13–25 yr) (Watanabe et al., 1992; Germann, 2011)	Perennial or annual (O'Clair and Lindstrom, 2000)	Perennial (O'Clair and Lindstrom, 2000; Mondragon and Mondragon, 2010)
Up to 30 m (Dayton, 1985)	Up to 36 m (Dayton, 1985; O'Clair and Lindstrom, 2000)	1 m (O'Clair and Lindstrom, 2000)	60 cm (O'Clair and Lindstrom, 2000)	<3 m (Mondragon and Mondragon, 2010)	Up to 3.5 m (O'Clair and Lindstrom, 2000)	1–2 m (O'Clair and Lindstrom, 2000; Mondragon and Mondragon, 2010)

Table continued

Table 1.—Continued.

	Alaria marginata	Costaria costata	Cymathaere triplicata	Hedophyllum sessile	Laminaria setchellii
Ecological adaptations & vulnerabilities	Fast-growing; mid-canopy species; abundant in pools (Luning, 1991; Mumford, 2007)	Opportunistic; disturbed areas; physiology depends on wave action; has potential to tolerate to warming ocean conditions (Dayton, 1985; Muth et al., 2019)	Tattered by July (Mondragon and Mondragon, 2010)	Competes poorly with other kelps; sea urchins avoid but chiton graze heavily; wave action determines growth form (Mondragon and Mondragon, 2010)	Has potential to tolerate to warming ocean conditions (Muth et al., 2019)
Human use[5]					
Food	Wakame substitute	Palatable, but low caloric value	Generally considered unpalatable (Druehl and Clarkston, 2016)	Palatable, but low caloric value	Kombu substitute (Jungwirth, 2019)
Nutrition	Source of vitamin A, vitamin B6, vitamin K, iodine, calcium, and potassium; contains more than 6% protein (O'Clair and Lindstrom, 2000; Mouritsen, 2013)			Source of trace minerals, complex carbohydrate and saccharides (O'Clair and Lindstrom, 2000)	
Medicine		Anti-inflammatory; thalassotherapy			
Industry		Previous source of potash salts (O'Clair and Lindstrom, 2000; Blasco, 2012; Mouritsen, 2013)			
Art					

[1]Abbreviations: SJDF = Strait of Juan de Fuca, SJI = San Juan Islands, PS = Puget Sound, OC = Outer Coast (Gabrielson et al., 2006; Mumford, 2007)
[2]The same species may have different temperature tolerances between regions. Recruitment refers to sporophyte production.
[3]Temperature is more limiting than nutrients (Muth et al., 2019). Optimum refers to maximum sporophyte density.
[4]Very little species-specific information is available, but salinity and turbidity are still crucial environmental factors to consider for sporophyte success. With the lack of numerical values, salinity and turbidity could be assessed by observing nearby use (for the possibility of sediment and freshwater influx), local currents/water flow, and the species' canopy form and depth.
[5]With the exception of a few commonly cultivated species, it is difficult to find species-specific human uses. Generally speaking, most kelp species can be put to the following uses: food, nutrition, medicine, art, and industry (food additive, animal feed, fertilizer, biofuel, etc.) (Tiwari and Troy, 2015).

Macrocystis pyrifera	Nereocystis luetkeana	Pleurophycus gardneri	Postelsia palmaeformis	Pterygophora californica	Saccharina latissima	Lessoniopsis littoralis
Adapted for exploitative competition for light/nutrients; susceptible to wave stress and grazing disturbance; has potential to tolerate to warming ocean conditions (Dayton, 1985; Muth et al., 2019)	Opportunistic; frequent disturbed areas; major source of carbon in inshore intertidal communities; sensitive to grazing; unlikely to tolerate to warming ocean conditions; vulnerable to exposure to petroleum products (Dayton, 1985; O'Clair and Lindstrom, 2000; Muth et al., 2019)	Where amphipod infestation occurs, warmer water temperatures may increase resilience to parasitism (Pfister and Betcher, 2018)	Forms dense stands; competes for space with mussels; unlikely to tolerate to warming ocean conditions (O'Clair and Lindstrom, 2000; Muth et al., 2019)	Long-lived, slow-growing; may form dense stands; adapted to physical stress (wave action, surface exposure) (Dayton, 1985; Mondragon and Mondragon, 2010)	Dusky Tegula snail and sea urchins are common grazers; young sporophytes sensitive to high light levels (O'Clair and Lindstrom, 2000)	Wide holdfast (Mondragon and Mondragon, 2010)
	Consumed pickled, or as a component of salsas, chips, soups, and other foods	Potential kombu substitute; low caloric value (Druehl, 1980; O'Clair and Lindstrom, 2000)	Eaten pickled, steamed or fresh		Commonly used in Asian cuisine; kombu/haidai substitute	
Source of many vitamins and minerals; low in tannins	Source of iodine		Source of fiber (Mondragon and Mondragon, 2010; Mouritsen, 2013)		Source of vitamin C, iodine, potassium, and calcium	Source of trace minerals and complex carbohydrates (Druehl and Clarkston, 2016)
	Mucilage may help treat burns					
Major source of alginate (O'Clair and Lindstrom, 2000; Mondragon and Mondragon, 2010)	Feed for mussels, filter feeders (O'Clair and Lindstrom, 2000, Mondragon and Mondragon, 2010; Mouritsen, 2013)				Potential use for bioremediation (Ahn et al., 1998; O'Clair and Lindstrom, 2000; Bruhn et al., 2016)	
				Stipes used by basket makers (Druehl and Clarkston, 2016)		

ble 3). Unfortunately, given the newness of the kelp mariculture industry in Washington (there are 2 operating open-water sugar kelp farms in Puget Sound and 12 proposed farms undergoing the permitting process as of September 2023[4]), there is very little locally specific research. We addressed some gaps by speaking to researchers and regulators with local mariculture expertise; but, in many cases, our judgment about the relative importance of each issue is based on the mix of people and communities, ecological threats, and regulatory concerns unique to the U.S. portion of the Salish Sea. As more data become available, a continuous evaluation of the relative importance and magnitude of each environmental impact— positive or negative—should occur. In the meantime, we have identified the following as high priority potential environmental impacts for the region: marine mammal entanglement, interactions between kelp farm crop and infrastructure with fish communities and forage fish spawning events, benthic shading and disturbance, and genetic effects on wild kelp populations.

Marine Mammal Entanglement

Placing physical structures in the water may disrupt or impede the movement of wildlife, particularly if farms are placed in important feeding, breeding, or migration areas. Though we could find no published reports of marine mammals entangled in sea-

[4]Chadsey, M. Personal commun., 26 Dec. 2023. Washington Sea Grant. 3716 Brooklyn Ave. NE, Seattle, WA 98105.

Table 2.—Socio-ecological considerations for kelp species native to the Salish Sea.

	Alaria marginata	Costaria costata	Cymathaere triplicata	Hedophyllum sessile	Laminaria setchellii	Macrocystis pyrifera
Cultivation method[1]						
Source	Wild sorus harvesting, sexual reproduction	Wild sorus harvesting, sexual reproduction				Wild sorus harvesting (parent stock must be adapted to farm site exposure levels), sexual or clonal reproduction
Culture	Culture tubes in nursery aquaria	Culture tubes in nursery aquaria				Inoculation lines in nursery aquaria
Farming	Sporophytes outplanted to floating long-lines (5 m depth) (Blasco, 2012). Blades also have been grown in aerated tanks	Sporophytes outplanted to floating long-lines (5 m depth) or floating raft; co-culture with *N. luetkeana* (Fu et al., 2010; Blasco, 2012)				Sporophytes outplanted to floating long-line (1.5–5 m depth, depending on season and epiphyte pressure); direct (free floating inoculation line) or indirect (nylon rope seeding); co-culture with *P. californica*; avoid siting near high Fe and NH_4^+ levels (O'Clair and Lindstrom, 2000; Gutierrez et al., 2006; Schiel and Foster, 2006; Westermeier et al., 2006; Macchiavello et al., 2010)
						Summer (Gutierrez et al., 2006; Macchiavello et al., 2010)
Sorus harvest[2]	Fall (Oct.) (Blasco, 2012)	From summer (Jun.) through late fall (Dec.), peak in Sept. (Maxell and Miller, 1996; Blasco, 2012)				
Outplanting[2]						Winter outplanting more successful than summer (Gutierrez et al., 2006; Macchiavello et al., 2010)

Nereocystis luetkeana	Pleurophycus gardneri	Postelsia palmaeformis	Pterygophora californica	Saccharina latissima	Lessoniopsis littoralis
Wild sorus harvesting, sexual reproduction	Wild sporophyte harvesting	Wild sorus harvesting		*Source:* wild sorus harvesting, sexual reproduction	
Culture tubes in nursery aquaria				Culture tubes in nursery aquaria	
Sporophytes directly transplanted to benthic substrate (e.g. rebar stakes); co-culture with *C. costata* and *S. latissima*; siting near eelgrass beds may increase habitat benefits (Merrill and Gillingham, 1991; Maxell and Miller, 1996; O'Clair and Lindstrom, 2000; Carney et al., 2005; Olson et al., 2019; Calloway et al., 2020)	Sporophytes transplanted to floating long-lines (2.5 m depth) (Germann, 2011)	Sori transplanted to cleared rocky substrate bounded by mussels; could be combined with intertidal mussel cultivation (Thompson et al., 2010; Druehl and Clarkston, 2016; Calloway et al., 2020)	Co-culture with *M. pyrifera*, but may struggle to compete for light (Watanabe et al., 1992)	Sporophytes outplanted to floating long-lines (5 m depth); co-culture with *N. luetkeana*; farm in exposed or semi-exposed areas to minimize biofouling (Blasco, 2012; Peteiro and Freire, 2013; Bruhn et al., 2016; PSRF, 2019)	
Late spring (May) through fall (Nov.), with peak production in late summer (Sept.) (Maxell and Miller, 1996)	Spring (Mar.) through fall (Oct./Nov.) (Germann, 2011)	Spores develop in late spring/early summer (O'Clair and Lindstrom, 2000; Thompson et al., 2010)		Fall (Oct.) (Blasco, 2012)	
Winter (Dec.-Feb.) (Calloway et al., 2020)					
		Spores develop in late spring and summer (O'Clair and Lindstrom, 2000; Thompson et al., 2010)		Winter (Jan.) (PSRF, 2019)	

Table continued

Table 2.—*Continued.*

	Alaria marginata	Costaria costata	Cymathaere triplicata	Hedophyllum sessile	Laminaria setchellii	Macrocystis pyrifera
Peak growth[2]	No discernable peak growth period (Blasco, 2012)	Spring (Mar.-May) (Maxell and Miller, 1996; Blasco, 2012)				Depends on outplanting time (perennial species) (Gutierrez et al., 2006; Macchiavello et al., 2010)
Sporophyte harvest[2]	Late spring (Jun), to prevent grazing from snails (Blasco, 2012)	Depending on grazing pressure and tattering, late spring (May) or mid-summer (June/July) (Maxell and Miller, 1996; O'Clair and Lindstrom, 2000; Blasco, 2012)	Late spring, after maximal blade development but before tattering (O'Clair and Lindstrom, 2000)			Late spring, before biofouling becomes a problem (Gutierrez et al., 2006)
(Wild) harvesting recommendations[3]	Cut vegetative blades no closer than 4" from the base; leave reproductive sporophylls intact (O'Clair and Lindstrom, 2000; Jungwirth, 2019)			Cut fronds no closer than 6" from holdfast (Jungwirth, 2019)	Cut blades no closer than 2" from the base (Jungwirth, 2019)	

[1] Some species have not yet been commercially cultivated. When available, literature on restoration mariculture and experimental transplants was consulted for species that have commercial potential.
[2] Limited species-specific information, especially for species that are not commercially cultivated.
[3] These are recommendations for wild harvesting that would apply if and only if a self-perpetuating farm is desired and/or permitted (as might be the case with restorative mariculture). Concerns about preserving wild genetic population diversity might, conversely, require complete adult sporophyte removal prior to sorus/spore production.

Nereocystis luetkeana	Pleurophycus gardneri	Postelsia palmaeformis	Pterygophora californica	Saccharina latissima	Lessoniopsis littoralis
Peak density and stipe growth in early summer (June); maximum blade growth in late summer (Aug./Sept.) (Maxell and Miller, 1996)	Rapid growth in winter (Dec.) until peak growth rate is reached in late spring (May) (Germann, 2011)	Late winter/early spring (Thompson et al., 2010)		Winter and spring; highest density reached in late spring (May-June) (Maxell and Miller, 1996; Blasco, 2012)	
Summer, to avoid epiphytes and blade erosion; fall (Oct.) to allow for re-growth (Maxell and Miller, 1996; O'Clair and Lindstrom, 2000; Luning and Mortensen, 2015)	Balding/senescence occurs in summer (June) and fall (Oct.) before regrowth (Germann, 2011)	Spring (Apr.-June) before spore production begins (Thompson et al., 2010)		Spring (Apr.-May), to reduce loss to blade erosion, ephiphytes, and grazers; timing affects the proportion of sugars, proteins, minerals (Blasco, 2012; Peterio and Freire, 2013; Luning and Mortensen, 2015; Bruhn et al., 2016; Sharma et al., 2018)	
Only distal ends of blades (>12 m from bulb) should be removed if continued growth is desired (O'Clair and Lindstrom, 2000; Luning and Mortensen, 2015; Jungwirth, 2019)	Breaks at the frond above the abscission zone are non-lethal; individuals live 3-6 years (O'Clair and Lindstrom, 2000; Pfister and Betcher, 2018)	Cut blades at least 2" from base, leaving 1-3" of grooved blade for regrowth (Druehl and Clarkston, 2016; Jungwirth, 2019)		Boiling after harvest can reduce iodine levels, which may be needed for use as food (Luning and Mortensen, 2015)	Harvest no more than 10% of individual plant (Jungwirth, 2019)

weed farms worldwide, the moorings associated with kelp cultivation pose a potential entanglement hazard, particularly for marine mammals (Langton et al., 2019). Gray whales, *Eschrichtius robustus*, seasonally occupy Puget Sound in the spring and summer, and humpback whale sightings in the Salish Sea have increased in recent years (Miller, 2020). Reported whale entanglements in fishing gear have been on the rise on the U.S. west coast in recent years, but are primarily associated with either pot fisheries for crab and fish trapping where a single, relatively slack line may connect traps to individual surface buoys, or to discarded nets from the fishing industry (Assink, 2019). In Puget Sound, whale entanglement in fishing gear is relatively uncommon—five humpback whales were reported entangled in pot fishery gear between 2010 and 2014, and between 2013 and 2017, three gray whales were reported entanglements in pot/trap fisheries (Assink, 2019). Southern resident killer whales have not been observed entangled in fishing gear, though one Bigg's (Transient) killer whale was briefly entangled in pot-fishing gear off the coast of Vancouver Island in 2018 (Haagen, 2018; WWF, 2021).

Additionally, several pinniped species travel to the Salish Sea, including California sea lions, *Zalophus californianus*; harbor seals, *Phoca vitulina*; and the Eastern distinct population of Steller sea lions, *Eumetopias jubatus* (NMFS, 2022a; 2022b; 2022c). Pinniped entanglement and ingestion of marine debris is typically associated with finfish mariculture, wild catch fisheries, and crab pot fisheries, with the primary culprit being discarded or lost rubber bands and plastic packing bands (Kemper et al., 2003; Raum-Suryan and Suryan, 2022). The rate of pinniped entanglement off of Washington's northern coast ranges from 0.41% to 2.13% of the observed population, but entanglement rates within the Salish Sea have not yet been reported (Allyn and Scordino, 2020; Raum-Suryan and Suryan, 2022).

Changes to Fish Communities and Habitat for Forage Fish

Artificial structures in temperate marine waters can change the distribution and habitat use of invertebrates and fish. Artificial reefs and derelict fishing gear are two examples of habitat changes that result in changes to fish use in the Salish Sea (West et al., 1994, Favaro et al., 2010). Kelp farm infrastructure—including anchor systems, ropes, and buoys—introduces substrates for invertebrates and wild seaweed to attach, and also likely attract fish and invertebrate communities that would otherwise not occupy these habitats. In local waters, this type of infrastructure can be used by rockfish; pile perch, *Damalichthys vacca*; striped seaperch, *Embiotoca lateralis*; and shiner perch, *Cymatogaster aggregata*; among other species. Juvenile salmonids, such as coho, *Oncorhynchus kisutch*, and Chinook are similarly attracted to habitat complexity in marine and estuarine waters (Shaffer et al., 2020), and may use

habitat near kelp farm infrastructure for foraging. Larval rockfish are often observed under free-floating algae, seagrass, and detached kelp (Shaffer et al., 1995), and kelp farms may provide similar habitat types, though potentially an ephemeral one given the seasonality of cultivation. A review by Theuerkauf et al. (2022) found that kelp farms had variable effects on marine communities, but generally greater fish abundance, mobile invertebrate abundance, and species richness were observed near kelp farms compared to reference sites.

Buoys and other kelp farm infrastructure on the surface are likely to attract fish-eating birds such as the double-crested cormorant, *Nannopterum auritum*. Similarly, fish predators—such as harbor seals, *P. vitulina*; California sea lions, *Z. californianus*; and river otters, *Lontra* canadensis—could be attracted to kelp farms to feed on fish associated with these habitats. While it is possible that predation on salmonids could increase in waters near kelp farms compared to sites without farming infrastructure, the seasonal growing period for the kelp species most likely to be cultivated, sugar kelp (late fall to early spring), is unlikely to overlap with much of the spring outmigration period for most Pacific salmon populations in the study area. The interaction between juvenile salmon and kelp farms would likely depend on the farm's proximity to areas of outmigration and the timing of farm operations, but this issue warrants additional research.

Additional fish and kelp farm interactions could occur. Pacific herring utilize marine vegetation, such as eelgrass, *Zostera* sp., and macroalgae, as a substrate to deposit their eggs (roe) (Penttila, 2017). Herring spawn-on-kelp fisheries are facilitated by harvesting wild kelp and staging it in known herring spawn locations just prior to or during the anticipated event, or capturing mature herring and holding them in pens until they deposit their eggs on the suspended kelp (Schweigert et al., 2018) Washington law permits the harvest of Giant kelp, *Macrocystis pyrifera*, used for herring roe on kelp fisheries (RCW 79.135.410). Herring roe-on-kelp fisheries last occurred in the U.S. portion of the Salish Sea in the mid-1990's, but these were closed due to a decline in the Cherry Point population of Pacific herring stock (Sandell et al., 2019). In 2020, cultivated kelp was used by herring during a spawning event at one farm in southeast Alaska, even though the farm was sited in an area without known historic spawning (Milne, 2020). Herring spawning on a cultivated kelp crop in Washington waters would require coordination with WDFW (which has jurisdiction over forage fish), delay crop harvest until the eggs hatch, and/or degrade product quality due to residual egg material.[5]

Benthic Shading and Disturbance

Benthic vegetation, wildlife, and seafloor structure could be disturbed by anchors or other moorings used in marine mariculture (Grebe et al., 2019). The dense growth of kelp along cultivation lines could prevent photosynthetic radiation (PAR) from reaching the seafloor, thus influencing the amount of light received by natural vegetation or phytoplankton below a farm, if grown in shallow waters (<30 m). Reduced light transmission in the nearshore can impact eelgrass growth and result in plant mortality and fragmentation (WDNR, 2015) depending on the characteristics and percent of shading (Jones and Stokes Associates, 2006; Lambert et al., 2021). However, additions to the seafloor have been found in other types of mariculture to act as artificial reefs and increase habitat for benthic organisms (Tallman and Forrester, 2007; Theuerkauf et al., 2022) and wild kelp habitat in close proximity to seagrass meadows can enhance seagrass ecosystem functionality and landscape connectivity (Olson et al., 2019), though the role of farmed kelp habitat has not yet been studied in this regard. In contrast, shellfish aquaculture interactions with submerged aquatic vegetation in Washington are an active area of both litigation (Laschever et al., 2020) and scientific research (e.g., Tallis et al., 2009). As our understanding of the effects of on-bottom and off-bottom shellfish culture on native submerged aquatic vegetation evolves, this may involve regulatory changes that are likely to affect seaweed cultivation, especially in the absence of research specific to the impacts of seaweed cultivation. Additional possible impacts to the benthos include the increase of detritus/organic matter, changes in infauna assemblages, and changes in oxygen levels and seawater chemistry (Walls et al., 2017).

Genetic Effects on Wild Kelp

The release of reproductive material from kelp farms into the natural environment is considered by some to be the highest-risk potential environmental impact associated with the expansion of macroalgae mariculture (Campbell et al., 2019). The cultivation of non-native kelp species could lead to the establishment of invasive species that compete with native species for space, light, and nutrients (Campbell et al., 2019) and is not currently permitted in Washington (WAC 220-370-220). Even the cultivation of selectively bred native species could, over time, affect wild populations, leading to greater genetic resemblance between farmed and wild populations and potentially resulting in a loss in genetic diversity, a higher potential for disease transmission, and/or a decline in ecosystem resilience (Buschmann et al., 2017; Grebe et al., 2019). The challenge of maintaining genetic diversity in cultivated marine organisms has been demonstrated in the Salish Sea in pinto abalone (Dimond et al., 2022), and Olympia oysters, (Camara and Vadopalas, 2009), two species which are cultivated in hatcheries for restoration/rewilding purposes, necessitating rigorous hatchery protocols for spawning and production of animals for outplant-

[5]Dionne P. Personal commun., 6 May 2022. Washington Department of Fish and Wildlife, 1111 Washington St. SE, Olympia, WA 98501.

Table 3.—Summary of the potential ecological impacts of cultivating kelp in the Salish Sea, in alphabetical order within each category of Local Relevance. Italics denote issues of particular concern for the region, based on conversations with local researchers, regulators and tribes. Local relevance was determined to be high if a category either had local data to support it and/or was ranked high as a concern in the literature, medium if there was no local data and it had a high or medium ranking in the literature as a concern but had a significant amount of uncertainty, and low if there was no local data and it was ranked medium or low in the literature as a concern and had a significant amount of uncertainty for its impact.

Ecological impact	Potential benefit	Potential concern	Economic considerations	Local relevance	Local data
Benthic/ Eelgrass	Farm anchors/moorings create habitat-forming 'reefs.' (Tallman and Forrester, 2007)	Crop blocks PAR-photosynthetic radiation, shading benthic environment below farm. *Farm anchors/moorings disturb benthic vegetation, wildlife and seafloor structure.* (Grebe et al., 2019)	Additional cost of lower-impact anchor systems.	High	Yes - Concern; Mumford, 2007; Tallis et al., 2009; Thom et al., 2011
Genetics	Assist in selection of high temperature-tolerant strains to withstand ocean warming	*Crop populates environment with reproductive material, influencing wild population diversity.* (Buschmann et al., 2017; Grebe et al., 2019)	Timing and extent of harvest affects release of reproductive material.[1]	High	N/A
Habitat	Crop provides habitat and refugia for other organisms. (Gentry et al., 2020)	Harvest removes habitat and refugia created by crop. (Wood et al., 2017; Grebe et al., 2019) Predation on fish attracted to farm infrastructure. *Forage fish spawn.*	Biofouling and epiphytes reduce crop value. (Peteiro and Freire, 2013; Langton et al., 2019) Utilization of crop by protected species delays harvest or reduces crop value.	High	N/A
	Farm infrastructure provides habitat and refugia for other organisms. (Gentry et al., 2020)	Disruption of wildlife movement and migration. *Entanglement hazards for marine mammals.* (Langton et al., 2019) Potential source of marine debris.[1]	Utilization of farm infrastructure by protected species delays harvest or reduces crop value. Total or partial loss of farm infrastructure and/or crop.	High	Yes - Benefit; Rensel and Forster, 2007
Water quality	Crop extracts nutrients, chemicals and metals from surrounding water column – mitigation of eutrophication, pollution, and HAB. (Imai et al., 2006; Kim et al., 2015; Augyte et al., 2017; Langton et al., 2019; Gentry et al., 2020) Crop absorbs carbon and may stabilize local pH – localized OA mitigation or value in the carbon offset market. (Duarte et al., 2017; Froehlich et al., 2019; Langton et al., 2019; Gentry et al., 2020)	--	Large-scale farms needed to achieve significant water quality benefits. (Gentry et al., 2020) Absorption of some pollutants by crop (i.e. heavy metals) may limit commercial uses.	High	Yes - Benefit; Peabody et al., 2020
Disease	--	Crop spreads diseases to native kelp population. (Langton et al., 2019)	Loss of crop from disease outbreak.	Medium	N/A
Nutrients/ Food Web Support	Detritus and particulates and DOM from crop provide food web support for other organisms, affecting the productivity of higher trophic levels. (Miller and Page, 2012) Nitrogen (nutrient) remediation – see Water Quality above	Particulate and dissolved organic carbon overloading. (Grebe et al., 2019) Competition with other organisms in nutrient-poor areas. (Wood et al., 2017; Grebe et al., 2019	Nutrients increase productivity of co-cultured species, such as shellfish. (Miller and Page, 2012) Timing and extent of harvest affects nutrient supplement.[1]	Medium	N/A
Hydrology	Farm infrastructure/crop dampens wave energy, reducing damage to shoreline. (Langton et al., 2019)	Farm infrastructure impacts natural movement of water.	Larger farms dampen water movement to the detriment of kelp growing in the middle of the farm	Low	N/A

[1]Davis J. Personal commun., 1 Jan. 2020. Blue Dot Sea Farms and School of Marine and Environmental Affairs, University of Washington, 3707 Brooklyn Ave. NE, Seattle WA 98105.

ing.[6] Given the genetic distinctiveness of bull kelp, within the southern Salish Sea, especially within the Puget Sound Main Basin and South Sound (Gierke, 2019), it is possible that other local kelp species have genetic structure that may be spatially and environmentally driven, and that a precautionary approach should be taken initially in managing cultivated stocks.

Farm Design Guidance

With the most locally relevant potential ecological impacts in mind, we now consider the second dimension of our farm design guidance: the kelp cultivation process. This guidance was informed by growers' manuals from seaweed farming in the North Atlantic (Table 4), peer-reviewed articles on experimental mariculture design, and conversations with experts in kelp farming and/or Salish Sea ecology. All information was verified with experts, and local information was prioritized when it was available. The goal of our initial farm design guidance is to maximize potential ecosystem benefits and minimize potential ecological harms without compromising commercial viability. A summary of best practices for kelp cultivation is presented in Table 5.

First, site selection is one of the most important factors for minimizing ecological harm, maximizing ecological benefit, and ensuring a healthy and productive kelp crop. Next, cultivation practices also influence ecosystem service provisioning and crop productivity. Our farm design guidance at the cultivation stage is for in situ cultivation in the marine environment only, and assumes routine monitoring and adequate farm maintenance. For information regarding the hatchery and nursery phases of cultivation, we recommend consulting the various growers' manuals listed in Table 4.

Finally, the harvesting stage can impact the surrounding marine environment, as well as crop health and value. Two additional stages of the farming cycle—processing and transport to market—are also important, but beyond the scope of this paper. Moving forward, selecting suitable sites, cultivation practices, and harvesting protocols for kelp mariculture would greatly benefit from the development of industry-specific marine spatial planning tools that integrate biophysical, social, economic, and logistical considerations.

Site Selection Considerations

Water Quality

Ensuring cultivated kelp will be grown in cold, nutrient-rich waters with plenty of light will maximize potential yields and lessen the risk of disease. It may also minimize the risk of cultivated kelp diverting nutrients from native vegetation, and could even provide the added benefit of absorbing effluent nutrients in the system. Bacterial contaminants have been detected on kelps (Barberi et al., 2020; Lovdal et al., 2021), so in the absence of understanding how kelp species absorb and retain these pathogens we recommend siting kelp farms in areas that meet current health requirements for shellfish cultivation, as well as avoiding areas with a history of diseased organisms (Bishop et al., 2021).

Additionally, seaweeds absorb heavy metals present in the water (Shams El-Din et al., 2014; Filippini et al., 2021, Hahn et al. 2022). Given the current absence of any federal or state laws or rules to regulate heavy metal content in seaweed products, we recommend following the requirements used at the national level or in the European Union (E.U.) (Barbier et al., 2019). Table 6 provides general suggestions for water quality metrics and human health requirements. As the domestic seaweed industry develops, additional research on bacterial and heavy metal contamination is essential for developing better-informed health standards (though see Hahn et al., 2022, for recent local analysis).

Studies in the E.U. have determined that *E. coli* requirements are unnecessary for the commercial seaweed industry (Barbier et al., 2019). Prospective farmers should look at current water quality conditions of a potential farm site, as well as seasonal variations and future projections. We also suggest cross-referencing available species-specific information, seen in Table 1, with site conditions for the best outcome. Farmers interested in cultivating kelp for non-consumptive commercial uses, or non-commercial uses (bioremediation, etc.) can consider a wider range of water quality metrics. Though post harvesting processes are beyond the scope of this paper, it should be noted that processing practices may be able to mitigate some of these water quality concerns and should be an area of continued investigation.

Water Flow

A steady current is essential for bringing in the necessary nutrients (e.g., dissolved nitrogen) for kelp to grow while also flushing the area of detritus (Freitag, 2017). Adequate water flow also distributes the dissolved or particulate organic carbon sloughed off the cultivated kelp throughout the system, providing a nutritional supplement to other areas and preventing a nutrient overload on site (Campbell et al., 2019). An ambient current of 10-40 cm/s is optimal, with a maximum of 100 cm/s at peak ebb/flood (~1/2–2 knots) to prevent crop, rope, or mooring damage. Prospective farmers should also consider extreme event (i.e., winter storm) flow rates, not just average flow rates, when designing infrastructure as the strongest storms occur in winter and possibly overlap with peak cultivation periods (see species specific information in Table 1).

Lastly, the selected crop species' current tolerance should correspond to the farm site's level of exposure (Table 1). There is also evidence that water flow tolerance varies by population for some species (i.e., *Macrocystis pyrifera*), so farmers should consider collecting source seed from areas with similar levels of exposure as their farm site (Gutierrez et al., 2006). Whether or not this holds true for oth-

[6]Eardley C. Personal commun., 26 Sept. 2022. Washington Department of Fish and Wildlife, 375 Hudson St, Olympia, WA 98501.

er kelp species is a question for future research.

Water Depth

Appropriate water depth is important from a business perspective to ensure full growth of cultivated kelp species. A sugar kelp farm could be sited in as little as 3.5 m of water,[7] however, many commercial manuals recommend siting farms in water 5.5 m or deeper (Merrill and Gillingham, 1991; Flavin et al., 2013). In some instances, depth is also a proxy for distance from the shoreline, and excessive distance from the shoreline presents several disadvantages. First, farms located further from shore could increase the likelihood of generating marine space use conflicts. Shoreline protection is also more likely to be provided if farms are located closer to shore, though additional research needs to be conducted on the optimal distance from farm to shore to maximize this potential ecosystem service. Finally, from a business perspective, maintenance, access, and transport costs for farmers increase with distance from the shore. However, would-be farmers must also consider how proximity to shoreline owners and their viewscapes might influence the social acceptance of their operations.

Appropriate water depth is also important from an ecological perspective. Selecting farm sites with moderate water depth helps avoid negative interactions with protected species and habitats within the Salish Sea. Though siting farms in water deeper than the photic zone (about >30 m MLLW) eliminates shading native marine vegetation, farms sited too far from shore in water deeper than 35 m increase the likelihood of humpback whale interactions.[8]

Table 4.—Kelp cultivation resources.

Bergdahl, J. C. 1990. Nori (*Porphyra* C.Ag.: Rhodophyta) mariculture research and technology transfer along the northeast Pacific coast., in: I. Akatsuka (Editor), *Introduction to Applied Phycology*, p. 519-551. SPB Acad. Publ., the Hague, Neth.

Canadian Kelp Resources, Ltd. The Canadian Kelp Resources Kelp Seed Production Manual. Rev. May 2015. Bamfield, B.C., V0R 1B0.

Flavin, K., N. Flavin, and B. Flahive. 2013. Kelp Farming Manual: A Guide to the Processes, Techniques, and Equipment for Farming Kelp in New England Waters. Ocean Approved, Biddeford, ME.

Freitag, G. 2017. Seaweed Farming in Alaska. Alaska Sea Grant Marine Advisory Program, Ketchikan, AK. ASG-63, 8 p.

Hurtado, A.Q. 2022. Genetic Resources for Farmed Seaweeds – Thematic Background Study. FAO, Rome. (https://doi. org/10.4060/cb7903en)

Merrill E. and D. M. Gillingham. 1991. Bull Kelp Cultivation Handbook. Applied Algal Research Co. P.O. Box 31104 Seattle, WA 98103.NCRI Publ. NCRI-T-91-011.

Barbier, M., B. Charrier, R. Araujo, S. L. Holdt, B. Jacquemin and C. Rebours. 2019. PEGASUS - PHYCOMORPH European Guidelines for a Sustainable Aquaculture of Seaweeds, COST Action FA1406, Roscoff, France.

Mooney-McAuley, K. M., M. D. Edwards, J. Champenois, and E. Gorman. 2016. Best Practice Guidelines for Seaweed Cultivation and Analysis, Public Output report of the EnAlage project, Swansea, Wales. 36 p.

Redmond, S., L. Green, C. Yarish, J. Kim, and C. Neefus. 2014. New England Seaweed Culture Handbook-Nursery Systems. Conn. Sea Grant, Groton, CT. CTSG-14-01. 92 p.

Rolin, C., R. Inkster, J. Laing, J. Hedges, L. McEvoy. 2016. Seaweed Cultivation Manual. Shetlands Seaweed Growers Project 2014-16. NAFC Marine Centre, Univ Highlands Islands, Gremista, Lerwick, Shetland, ZE1 0PX.

Visch, W. 2019. Sustainable Kelp Aquaculture in Sweden. Ph.D. Thesis, Univ. Gothenburg, Goteborg, Sweden.

Native Aquatic Vegetation

Commercial kelp farms should avoid locations over or near significant stands of native marine vegetation, such as macroalgae or eelgrass beds, and minimize overlap with smaller stands of such vegetation. Current DNR regulations require shellfish farms to provide a 25 ft buffer between native vegetation and the farm perimeter and in the absence of seaweed farm-specific regulations, the same guidance applies (ARD, 2014). However, the interaction between vegetation and shellfish aquaculture, and the impact of overwater structure shading on macroalgae (Lambert et al., 2021), show inconsistent results and is an active area of research, thus future results may justify additional review of this requirement, for both seaweed and shellfish farms.

Observing the location patterns of native vegetation is not only important for regulatory compliance but also may help guide farmers in best mimicking natural conditions for their crop. For example, wild kelp species typically grow on substrates that enable holdfast development such as rocky bottoms and unconsolidated substrates with rock/cobble. Farm placement above these types of seafloor substrate, as long as it is bare of native vegetation, may allow the farm to better provide similar habitat and other ecosystem services.

Marine Mammals

To reduce the likelihood of gear interaction, entanglement, and behavioral disruption of large whales, farms should be sited where there is minimal overlap with known large-whale feeding areas and migration corridors (Miller, 2020). For instance, most gray whales within the study area seasonally feed within the North Puget Sound near the Snohomish River delta (Calambokidis et al., 2015). Humpback whales, whose population numbers are increasing, are also known to seasonally feed throughout Puget Sound (Miller, 2020).

Forage Fish Spawning

Prospective kelp farmers should consider avoiding known and consistent herring spawning areas when siting farms. Research is underway in Southeast Alaska regarding best practices to avoid spawning events on cultivated kelp, and findings of this work should be incorporated into Salish Sea

[7]Davis J. Personal commun., 1 Jan. 2020. Blue Dot Sea Farms and School of Marine and Environmental Affairs, University of Washington, 3707 Brooklyn Ave. NE, Seattle, WA 98105.

[8]Lawson, D. Personal commun., 13 Apr. 2020. NMFS, West Coast Regional Office, Long Beach Branch, 501 W Ocean Blvd, Long Beach, CA 90802.

Table 5.—Summary table of kelp farm design guidance.

Farming stage	Key question(s)	Farm design considerations/suggestions	Related potential environmental impact(s)
Site Selection	Where to site? Which species?	Ensure site water quality meets growth requirements for kelp and contaminant requirements for human health.	Water quality, nutrients, disease
		Ensure adequate water flow through the farm.	Nutrients, hydrology
		Site farms in moderate water depth (5.5–35m).	Benthic/eelgrass, habitat
		Minimize or avoid overlap with native aquatic vegetation.	Benthic/eelgrass
		Avoid large whale (i.e., gray whale) feeding areas and migration corridors.	Habitat
		Avoid areas with a prior history of forage fish spawning events	Habitat/species
Cultivation	Which species? How much seed? When to outplant? How much crop? Which materials? Which method(s)?	Cultivate only native species, and multiple species when possible.	Habitat, nutrients, water quality, disease, genetics
		Collect small amounts of seed stock from within the same oceanographic basin.	Genetics
		Use cable anchor lines, taut longlines with a weight of ⅜ to 1 ¾ inches.	Habitat/entanglement
		Use embedded anchors for moorings.	Benthic/eelgrass
		Orient longlines parallel to prevailing currents, minimize the use of plastic gear, and ensure proper gear maintenance and disposal.	Habitat
		Place longlines 2–5m deep, 1.8-4.5m apart horizontally and at least 1m apart vertically.	Habitat, nutrients, water quality, hydrology, disease
Harvesting	When to harvest? Which parts? How much crop?	Harvest crop in at peak biomass/ upon maturity of the product.	Genetics, habitat
		Remove the entire crop and all longlines during harvesting.	Genetics, habitat, disease

kelp farms, as appropriate (NMFS, 2021). If herring spawn on farmed kelp occurs, farmers should consult with WDFW and delay harvest until hatching occurs.[5]

Cultivation Considerations

Native Species

Protecting the genetic diversity of wild populations is essential for the conservation and recovery of kelp in Puget Sound (Calloway et al., 2020). To protect the genetic diversity of wild kelp populations and to ensure the crop is relatively adapted to local ecological conditions, we suggest collecting seed stock from wild kelp within the same oceanographic basin as the farm site (Fig. 1) or as close as possible. The sori (reproductive tissue) from the equivalent of 5 to 15 sugar kelp sporophytes (adult "plant" phase) provide enough gametophytes to seed up to 20 km of line[9], though in Alaska, another state with a growing kelp mariculture industry, it is currently recommending that sori be collected from at least 50 individual plants to best protect genetic diversity (Gruenthal and Habicht, 2022).

With wild, local broodstock collection as one part of a risk averse pathway, it is important to acknowledge that the development of cultivars (selectively bred varieties), could benefit farmers by allowing them to select for traits that maximize yield, resilience, and profitability (and/or ecosystem services), enable a more efficient use of marine space by allowing greater crop production within a smaller farm footprint, and allow for better adaptation to climate change (Goecke, 2020). Investigations are underway in the United States, Europe, Australia, and South Korea, and under scrutiny of policy makers to selectively breed kelp in an ecosystem-responsible manner (Goecke et al., 2020; Mao et al., 2020; Azra et al., 2022; Huang et al., 2022; Vissers et al., 2023). Genetic risk tools, such as the Offshore Mariculture Escapes Genetics Assessment model, could be adapted to seaweed culture to predict and evaluate the risk of invasions and introgressions from cultured macroalgae on natural algal populations. In the future, farmers or third-party entities might develop cultivars, seed banks, or other seed stock programs derived from native species.[9] Sterile cultivars for in situ commercial kelp mariculture is already in development elsewhere (Vissers et al., 2023), and would enable more efficient crop production while reducing the risk of

[9]Dobbins P. Personal comms., 5 May 2020. World Wildlife Fund, 1250 24th St NW, Washington, DC 20037

Table 6.—Optimal water quality elements for growing kelp and crop metal/mineral limits for ensuring human health (U.S. requirements unless otherwise noted).

Indicator	Range
Temperature	10-15° C, Max 18° C
Nutrients	1-20 (Optimal >10) µmol/L NO_3^-, >0.3 µM PO_4^{3-}
Salinity	Minimum 24 ppt
Light Availability at kelp depth	1,000 – 2,000 lumens m^{-2} (20-40 $\mu Em^{-2}s^{-1}$)
Fecal coliform	< 30-43 FC/100mL (WA Dept. of Health)
Lead	<10 mg kg-1 DM, ppm
Cadmium	<0.5-3 kg-1 DM, ppm (from France/EU)
Mercury	<0.1 kg-1 DM, ppm (from France/EU)
Inorganic arsenic	<3.0 kg-1 DM, ppm
Iodine	<5,000 kg-1 DM, ppm

genetically altering wild kelp populations. In addition to the ecological considerations discussed here, the privatization of developed cultivars and related questions of intellectual property rights are likely to be an important economic/social concern similar to seed ownership in terrestrial agriculture. In Table 2 we provide a list of native kelp species with commercial potential.

We also encourage cultivating a diversity of kelp species within farms, as it would likely enhance both ecosystem services and economic returns. In terrestrial settings, crop biodiversity increases the potential number of commercial products and may improve resilience to disease, parasites, predation, or other unexpected natural events (Lin, 2011). Simultaneously growing species with multiple canopy forms (prostrate, stipitate, floating) uses the entire vertical dimensionality of the farm, better mimicking the habitat created by wild kelp beds and maximizing potential growing space, though also adding complexity to farming methods and timeline. A possible combination of species that have complementary canopy forms and have been experimented with include bull kelp, ribbon/winged kelp, *Alaria marginata*, and sugar kelp (Table 2). Farmers could also extend the growing season by sequencing kelp species with different growth and harvesting seasons. This would maintain kelp biomass and associated ecosystem services in the marine environment for a longer time period.

Importantly, the feasibility of increasing the spatial and temporal biodiversity of kelp farms will depend on the other uses of marine space occurring at or near the farm site. Commercial and subsistence tribal fishing activities, vessel traffic, marine mammal migrations, and/or forage fishing spawning events may preclude having kelp longlines in the water for an extended period of the year. Prospective farmers interested in kelp polyculture should keep these limitations in mind when selecting potential farm sites and communicating with other users of the same marine space.

Anchor Lines and Longlines

Kelp is commonly cultivated on horizontal lines suspended in the water column, either as separated longlines or connected in a grid-like pattern (Hurtado, 2022). These longlines are attached to benthic moorings by anchor lines. A similar setup may use vertical dropper lines instead, one of many potential variations in design.

To minimize the risk of marine mammal entanglement, longlines should be between 3/8" and 1¾" diameter and kept as tight as possible for all tidal conditions. Farmers should check the tautness of longlines frequently and do a thorough inspection of moorings every year and after storm events. Other suggestions to minimize risk of entanglement include using cables for the anchor lines and potentially using breakaway toggles or links at multiple points on the cultivation infrastructure. Additional research, however, is needed to test the feasibility (and design) of using breakaway toggles,[8] to assess the potential for loss of gear.

Embedment Anchors

Although kelp farmers in other regions routinely use concrete moorings[10], DNR currently discourages the use of concrete blocks as moorings to minimize benthic disturbance (WDNR, 2018). Kelp farmers should mostly consider the use of embedded anchors for moorings; however, with additional research, the use of pre-existing concrete moorings could be a viable option given the potential habitat benefit of providing substrate for wild kelp (Tallman and Forrester, 2007). Concrete mooring structures may also be more effective for alternative cultivation techniques, such as those for bull kelp. Concrete moorings located within the photic zone have been effectively used in local bull kelp restoration efforts in Puget Sound (Puget Sound Restoration Fund, 2019).

Longline Orientation

Like many commercial activities occurring on the water, kelp mariculture can be a source of marine debris, which poses a hazard to local marine flora and fauna (Feng et al., 2020). Farmers should minimize the use of plastic materials, including polylines which can erode and shed microplastics into the environment (Napper et al., 2022). They should also ensure the collection and proper disposal of waste materials, excess line, and other debris consistent with applicable local, state, and federal regulations.

To avoid excess wear and tear to equipment in the first place, farms should be oriented so longlines run

[10]Piconi, P. Personal comms., 5 Feb. 2020, at Washington Sea Grant Seaweed Farming Workshop. Island Institute, 386 Main St., P.O. Box 648, Rockland, ME 04841.

parallel with the prevailing current. All gear placed in water should be labeled with a unique identifier in the case of loss. Research on the use of recycled carbon fiber cables for grow lines and other gear is underway at one commercial seaweed farm in Washington. Using cables that are stiffer than commercially available lines, have a very long lifespan, and do not shed microplastics through chafe and exposure to UV radiation could contribute to more sustainable farming practices.

Longline Spacing

Determining the appropriate farm scale and density for any particular site requires balancing various tradeoffs. In economic and ecological terms, farm scale and density influence crop productivity, which in turn affects profits, the provisioning of ecosystem services, and the risk of disease, parasites, and predation. In practical terms, longline density will in part depend on vessel size and the ease of movement through the farm. Many of the specifics will depend on site-specific features of farms, including nutrient availability, current strength, tidal regimes, and water depth, as well as adjacent land and water uses.

With these caveats in mind, we suggest spacing longlines 1–4.5 m apart along the horizontal axis (Flavin et al., 2013).[9,11] If growing multiple species with different canopy forms, a grower may want to use stacked vertical lines to maximize farm space. In this case, farmers should space vertical lines about 1 m apart throughout the depth of the water column, starting 2–5 m below the surface depending on the canopy form of the cultivated species. More locally-based research should be conducted to study the effects of longline density on crop productivity, water quality, habitat provisioning, and other ecosystem services to better inform these preliminary suggestions.

Farm scale in the Salish Sea will likely be more constrained by socio-economic factors than by ecological factors, with economic viability determining the lower limit and marine space use conflicts setting the upper limit. Few studies have quantified how environmental impacts scale with farm size, but one study on the west coast of Sweden found no ecological costs associated with sugar kelp farms up to 2 hectares (~5 acres) using multiple configurations of longlines (50,200 m longlines vs. 24,400 m longlines) (Visch, 2019). Research on the environmental impact of kelp farms at larger scales should be a high priority.

Harvesting Considerations

Harvest timing

After peak growth (typically in the winter for sugar kelp) kelp will continue to accumulate biomass throughout the next couple of months. Changes in water temperatures and surface nitrogen availability, such as warming during the summer and increased stratification, increase the risk of grazing, epiphytic biofouling, disease, and blade degradation. Additionally, reproduction generally occurs following the peak growth period. To avoid these issues, which could reduce crop value, increase processing time, or spread disease or genetic material to the wild stock, it is best to harvest the crop after the peak growth period. Consult Table 2 for species-specific harvesting seasons.

Crop and Longline Removal

When harvesting an annual species, we suggest removing the entirety of all cultivated plants on the longlines. For perennial species, remove as much reproductive material as possible. These measures minimize the risk of cultivated species spreading reproductive material or diseases into the natural environment and impacting the genetics or health of the wild stock. Where feasible with farm operations and growing methods, we recommend the removal of all longlines after harvest to minimize biofouling, the risk of wildlife entanglements, and to avoid marine space use conflicts—namely the interference with tribal fisheries.

Future research to inform policy development should investigate the tradeoffs involved in allowing some crop and/or gear to remain in the water for extended time periods. For example, cultivating a diversity of kelp species in rotation throughout the year, or coppicing (partially harvesting and allowing the kelp to regrow for a second harvest) might maximize both ecosystem services and profits. However, these potential benefits would need to be weighed against the risk of wildlife entanglement at the farm site, and farmers would need to consult with tribes and other marine space users in the area. Similarly, in some areas the habitat and nutrient provisioning benefits of leaving non-reproductive crop tissue and farming gear in the water may outweigh the potential risk of gene flow from domestic to wild kelp populations.

Discussion

Key Challenges

While we have outlined practical and precautionary ways prospective kelp farmers could design and operate within the U.S. portion of the Salish Sea, the greatest challenges to developing a new industry may be social acceptance and the ability to coexist with other pre-existing water and land uses adjacent to the site, including: shipping lanes, fishing and other vessel traffic, viewsheds of private waterfront property owners, tribal U and A's and reservations, marine protected areas and aquatic reserves, proximity to industrial or wastewater treatment facilities, and proximity to processing facilities.

Coordination with tribes is critical to gaining permits to operate a kelp farm. Many tribes take an active role in reviewing local, state, and federal permitting proposals by providing feedback to regulatory entities. Invariably, the infrastructure needed for a kelp farm, such as anchors, lines, and buoys, requires authorization by the Army Corp of En-

[11]Grebe, G. Personal comms., 5 Nov. 2019. Marine Science Center, Univ. of New England, 11 Hills Beach Road, Biddeford, ME 04005.

gineers (ACOE) under Section 10 of the Federal Rivers and Harbors Act. Tribes may volitionally review draft permit applications and raise objections directly to the ACOE. With input from the tribe(s), the ACOE must determine whether proposed projects would have more than a de minimis impact to treaty rights, and if they do, the ACOE will not issue permits. The ACOE also consults with the National Marine Fisheries Service and the U.S. Fish and Wildlife Service to assess the actions effects on ESA listed species and designated critical habitat. On a state government level, seaweed farming projects that are proposed in state-owned aquatic lands require authorization from the Washington Department of Natural Resources via a lease agreement. On a local level, the Washington State Shoreline Management Act required jurisdictions to develop Shoreline Master Programs (SMP's). The Act calls for each jurisdiction to develop an SMP that outlines plans to 1) balance and integrate interests of local citizens; 2) address shoreline conditions; 3) consider and influence planning regulations for adjacent lands; and 4) classify shoreline segments into environmental designations (WECY, 2012). SMP's must strive for no net loss of ecological function while still promoting the public use and development of county and city shorelines, using the "best available science" to issue specific guidelines (RCW 90.58). Because seaweed farming is a novel activity in Washington waters, there is virtually no up-front local, state, or Federal guidance to ease permitting for proposed farms.

Depending on their location, kelp farm infrastructure could interfere with pot, dive, gillnet, beach seine, and drift net fisheries targeting Pacific salmon, Dungeness crab, *Metacarcinus magister*, spot and dock shrimp, *Pandalus platyceros* and *Pandalus danae*, and geoduck, *Panopea generosa*, and potentially other fisheries. Early engagement with potentially affected tribes, prior to submitting permits, is critical to developing collaborative approaches to locating and operating kelp farms in local waters.

Specifically, we recommend considering locations that avoid or minimize fisheries conflicts. These might include areas with few other targeted resources at the site, or areas that are already "developed" due to the presence of nearshore infrastructure (docks and piers) or activities (shellfish mariculture) that impact fisheries access. Depending on the season, the removal of kelp farm infrastructure following the harvest season, such as lines and buoys, has the potential to partially alleviate conflicts with tribal fisheries (summer), salmon fisheries (late fall), and crab fisheries (winter) but will inevitably still overlap with some fisheries.

Similarly, tribal concerns about environmental degradation from proposed projects are taken under consideration by multiple local and state regulators as well as the ACOE and other federal government agencies with a regulatory role. Legal rulings have also deemed the preservation of fish habitat by state and federal regulators central to upholding treaty rights (NWIFC, 2014). The Western Washington Treaty Tribes have long expressed concern about the loss and degradation of habitats that support treaty fish and shellfish, which they attribute to "a lack of coordinated federal leadership, a failure to exercise authorities, and the disparate application of salmon conservation measures" (Treaty Indian Tribes in Western Washington, 2011). Many tribes have research and conservation programs aimed at restoring habitat functions in the rivers and estuaries that flow into the Salish Sea. They also support marine shoreline and nearshore protection and implement restoration actions such as removal of shoreline armoring and creosote logs, and conducting native oyster restoration projects. For these same reasons, an industry that is potentially ecologically beneficial, such as kelp mariculture, may be of interest to tribal governments.

Developing a Monitoring Framework

Monitoring and evaluating outcomes is a key element of any adaptive management process. The absence of regionally specific data on the ecological impacts or benefits of kelp farming generates uncertainty for resource managers, farmers, and the general public. In light of this uncertainty, further development of the kelp mariculture industry requires the development of a transparent and collaborative monitoring framework based on the best available science. We recognize that monitoring entails a significant burden of both time and resources, but an investment in monitoring can yield valuable information about ecological impacts and economic efficiency that might inform specific regulatory requirements and broader public perceptions related to kelp farming. We also recognize the difference between monitoring that is required by regulators, and monitoring that farmers voluntarily conduct to improve their own farm operations or social license.

Key Process Considerations

Concerns about information-gathering and decision-making processes often underpin social license challenges for marine industries and a lack of trust in regulatory agencies (Uffman-Kirsch et al., 2020). Given the inherent social and political complexity of siting private commercial activities within public waters and the particular challenges of balancing multiple uses of marine space in the Salish Sea (Eichenberg and Vestal, 1992; Marston, 1996), questions about the who, why, and how of monitoring are perhaps even more important than questions about what to monitor. A monitoring framework for kelp mariculture in the region should emerge through a participatory process that engages researchers and regulators, fosters industry accountability, respects tribal sovereignty, and ensures equity and access—particularly for historically excluded groups (LaFrance and Nich-

ols, 2008; Mertens, 2008). The framework should also identify roles and responsibilities for implementing and funding each aspect of monitoring, craft rules about collecting, analyzing, storing, and sharing monitoring data, and consider how monitoring data will be used to inform management recommendations. For example, the framework should distinguish between minimum required monitoring for regulatory and permitting purposes versus voluntary monitoring to assess industry objectives or address community concerns. For guiding principles and key questions to consider when establishing a monitoring framework for kelp mariculture, refer to Table 7. This topic, as well as larger policy recommendations, is addressed in greater detail by The Nature Conservancy's Principles of Restorative Aquaculture (TNC, 2021).

Key Elements to Monitor

Based on the potential environmental impacts we previously identified and the remaining research gaps and uncertainties, we recommend, at a minimum, monitoring the following four elements: 1) water quality, 2) species assemblages, 3) hydrology, and 4) kelp genetics. Within each of these elements, some aspects will be more important from a regulatory perspective (i.e., required), while others will matter more from an industry or community perspective (i.e., voluntary). For required monitoring, farmers should work with regulators in advance to determine the bounds and scope of monitoring activities at their farm and come to a written agreement. As elaborated below, more comprehensive monitoring should be pursued by partnerships among local, state, and federal permitting agencies, NGO's, and academic institutions to develop monitoring regimes appropriately scaled for individual sites and operations.

First, measuring changes from baseline water quality in and around kelp farms will be important for ensuring crop health and product safety, monitoring impacts on ambient nutrient levels, and providing empirical evidence of the water quality improvements expected from kelp mariculture. We suggest required monitoring for water quality parameters related to potential ecological impacts (e.g., dissolved inorganic nitrogen) and public health (e.g., heavy metals, pathogenic bacteria (Lovdal et al., 2021)), and voluntary monitoring for parameters indicating potential benefits of public and/or academic interest (e.g., pH, pCO_2). Potential parameters of water quality to measure are listed in Table 8.

Second, to evaluate the habitat impacts of kelp mariculture, monitoring should include surveys of species assemblages in, around, and below farm sites (reviewed in depth by Corrigan et al., 2022). Species presence and abundance on farm sites could be measured in an absolute sense, or in comparison to baseline conditions or to adjacent areas. Methods might include dive surveys, underwater cameras, eDNA, benthic grabs, or other methods. The most appropriate methods will likely be informed by the intended purpose and audience for monitoring. For example, underwater cameras could yield footage that is valuable for enhancing social license and communicating with the public, while simultaneously providing required and/or useful information to regulators and/or researchers. We also recommend mandatory reporting of any interactions with protected species, such as marine mammal presence or entanglement, forage fish spawning, salmon migration, or changes in submerged aquatic vegetation.

Farm hydrology monitoring should be considered by farmers, including collecting baseline and ongoing data on water depth, wave exposure/velocity, tidal gauge, and substrate to document the potential hydrological impacts of kelp mariculture, as well as better understand and track farm conditions. Regulators and researchers may consider additional data collection, in conjunction with the above, on shoreline slope, exposure, and substrate. These data could clarify the potential for kelp farms to protect the shoreline from storms or vessel traffic. Finally, as a complement to the wild kelp bed monitoring proposed in the Kelp Conservation and Recovery Plan (Calloway et al., 2020), at a minimum kelp farmers should be required to document the location of sorus material they use for seed. It might also be helpful for researchers, regulators, and farmers to maintain a seed bank or preserved DNA of harvested sori and kelp crops from different locations to document existing genetic diversity and to track any changes in genotype or phenotype over time (e.g., disease susceptibility, temperature tolerance, etc.). Ideally, this would allow regulators to monitor potential impacts to wild kelp beds as well as facilitate the development of domesticated seed stock by farmers. We recommend seed source documentation as the first priority in developing biosecurity practices for this emerging industry. Biosecurity measures are a critical component of successful mariculture as they aim to prevent, identify, control, and minimize the spread of pathogen and pest infestations among farms, and prevent impacts on surrounding ecosystems. Practices such as farm hygiene, inspection, laboratory testing, quarantine, and rigorous permitting regulations and procedures for import and transplant are common in shellfish and finfish aquaculture and may be similarly developed for seaweed cultivation as a better understanding of pest and pathogen risks for these species emerge.

Other Future Considerations

Co-culture and Polyculture

Interest is increasing in various forms of kelp co-culture and polyculture, which is the cultivation of several commercial marine species within the same structure or farm site. Integrated multi-trophic aquaculture (IMTA) in particular has been proposed as an innovative "ecosystem approach" to maximizing ecological benefits and diversifying commercial opportunities in mariculture (Langton et al., 2019). IMTA systems cultivate primary producers (e.g., seaweeds) to re-

Table 7.—Guiding principles and questions to consider for the development of a kelp mariculture monitoring framework.

Framework component	Guiding principles	Key questions
Participation	Diversity Inclusion Power	Which entities and/or individuals have a stake in industry development? (industry, researchers, tribes, NGOs) Who might be affected by industry development? (residents, workers, recreational users, tribes) Which groups have traditionally been excluded? (groups historically excluded based on race, ethnicity, language, class, education, etc.) Which individuals/organizations have decision-making power in the regulatory and proprietary process? (federal and state regulators, local regulators, tribes)
Roles	Cost/benefits Conflicts of interest Equity	Who is responsible for collecting the data? Who is responsible for analyzing the data? Who is responsible for storing the data? How will these different monitoring activities be funded, and by whom?
Protocols	Access Information Resources	How should monitoring data be collected and analyzed? Knowledge systems (e.g. Western science, Indigenous science, citizen science) Methods (e.g. quantitative tools, indicators and metrics; qualitative tools, indicators and metrics) Frequency (e.g. continuously, monthly, seasonally, annually) Spatial scale (e.g. acre, farm, basin, etc.) How will the results of monitoring be shared, and with whom (e.g. industry, regulators, tribes, researchers, the public)? Who should have direct access to monitoring data (e.g. farmers, regulators, tribes, researchers)? What are the allowable changes from background or control sites that are allowed? What accuracy and precision is required and the required sampling design? Are these changes attributable to the farm or other causes? How will this fit into a "no net loss" requirement?
Recommendations	Adaptive management	How will monitoring data be used to inform management decisions?

cycle the nitrogenous wastes of consumer species (e.g., finfish and shellfish) while also providing particulate organic matter (POM) to filter feeders (e.g., shellfish) (Miller and Page, 2012; Langton et al., 2019). By utilizing the nutrients contained in the waste products of one species as a feed input for another, IMTA systems mitigate some of the environmental impacts of finfish mariculture and could boost farm profitability by reducing inputs and increasing crop biomass. Co-culture has previously been particularly successful with abalone and non-kelp macroalgae species such as dulse and sometimes *Ulva lactuca* (Evans and Langdon, 2000; Demetropoulos and Langdon, 2004; Langdon et al., 2004; Hamilton et al., 2022), in this case co-culturing the seaweed as direct feed for the invertebrate species as well as nitrogenous waste recycling.

Within temperate regions, such as the Salish Sea, the co-culture of kelp and bivalves may be a good entry point into kelp farming, particularly for already established shellfish farmers (Granada et al., 2018; Langton et al., 2019). Although bivalves are typically grown in shallower water than is recommended for most kelp species, sugar kelp can be cultivated in as little as 1.5 ft of water (Greenwave, 2021; Jiang et al., 2022), making this species a possible candidate for co-culture with bivalves like oysters. Co-culturing is often done on the same line, or on separate lines in close proximity to each other, thus most of our farm design guidance could also apply to co-culture systems. With the inclusion of kelp/seaweed IMTA systems in the new nationwide permit for seaweed mariculture (ACOE, 2021), co-culture systems are a worthwhile area of experimentation and further research in the Salish Sea.

Hybrid Models of Mariculture

Though we have focused on commercial kelp mariculture in this paper, hybrid industry models that combine commercial mariculture with restoration, bioremediation, or mitigation warrant further consideration (see Mizuta et al., 2022 for definitions of aquaculture for environmental purposes). For example, as part of a commercial/restoration hybrid approach, kelp farms could be established in areas where kelp loss has occurred and farmers could leave some kelp crop with reproductive material on the lines to help repopulate kelp beds. Balancing commercial and restoration goals in this manner, however, would require careful research on the population genetics of this region's native kelp, the establishment of a seed stock distribution program, and thoughtful partnership between farmers and

Table 8.—Potential water quality indicators to include in a kelp farming monitoring framework.

Purpose	Parameter
Crop health	Temperature Salinity DIN (nitrate, nitrite, ammonia) Phosphorus Turbidity
Human health	Fecal coliform Lead Cadmium Mercury Inorganic arsenic Iodine *Vibrio* spp. *Bacillus* spp. *Aeromonas* spp.
Ecological health	Dissolved inorganic nutrients Dissolved oxygen Chlorophyll-a pH pCO_2

Figure 2.—Schematic of aquatic phase for floating long-line kelp mariculture with the marine interactions of most significance for the Salish Sea indicated: tribal subsistence and commercial fishing, shipping, benthic vegetation (such as eelgrass), forage fish habitat and spawning events, marine mammal entanglement risk, and wild stock genetic diversity.

conservation agencies (Mizuta et al., 2022).

Similarly, kelp mariculture could be sited in areas with excessive nutrients or other water quality issues to aid in bioremediation while simultaneously boosting crop production. The nature of water quality issues at specific sites could limit use of the kelp crop if the targets for bioremediation are harmful to human or animal health. Thus, both careful coordination between farmers and regulators working in environmental and public health and more diverse markets for local kelp products would be necessary (The Nature Conservancy, 2021).

Payment for Ecological Services

Previous studies have modeled the potential effects of seaweed mariculture on hypoxic conditions, ocean acidification, carbon sequestration (Froehlich et al., 2019), and nutrient pollution (Racine et al., 2021) under different scenarios. However, there is significant criticism of the idea that kelp farms could serve as a climate "fix," particularly due to the lack of evidence that seaweed could be used as a true carbon "sink" or method for sequestration (Troell et al., 2022), challenging hopes for carbon offsets funds as a potential economic framework for the industry. It is likely more beneficial to utilize a local monitoring framework (such as outlined above) for the industry to build stronger empirical evidence for ecological services (listed in the Potential Ecological Impacts section) provided by kelp farming, and explore the value of those services for potential future payment or subsidies. Understanding how these services may extend beyond the local marine environment, such as offsetting carbon through product replacements for animal feed, fertilizers, bioenergy, etc. (Troell et al., 2022) would be an additional important consideration.

Land-based Culture Operations

Although this paper focuses primarily on open-water culture, land-based culture merits mention, as this approach offers some specific advantages, and avoids a few important potential challenges with kelp farming in open water, namely cetacean entanglement and concerns for genetic effects on local populations. Land-based systems, lacking ropes and other submerged gear in the marine environment, do not pose the same hazards to marine mammals of open-water culture, and allow for sterile, filtered effluent, minimizing genetic contamination and other risks. Including land-based culture among the potential continuum of intensive and extensive kelp farming and harvest methods could also reduce pressure on wild kelp beds and their associated ecosystems.

Among some of the direct advantages, land-based operations allow for more controlled growing conditions; they may in some cases enable year-round production, high control over quality and product value, enable alteration to morphology and size, can limit or eliminate biofouling, and could avoid introduction of chemical contaminants through influent water treatment (Hafting et al., 2012; Mata et al., 2016; Pereira et al., 2024). Entirely land-based growout operations can also use a variety of culture techniques through harvest, focus on rapid growth, often emphasizing asexual, vegetative reproduction. Land-based operations include hatcheries, which could potentially be used to produce sterile cultivars for later grow-out in more traditional open water farms (Goecke et al., 2020; Vissers et al., 2023). A land-based operation can treat effluent to prevent release of products that could lead to wild reproduction and genetic contamination, decreasing the risk of overwhelming wild stock compositions and diversity reduction or hybridization with, and introgression into, local populations.

The inherently more intensive nature of land-based culture is responsi-

ble not only for many of its advantages, but also incurs specific costs and responsibilities. Additional costs of potentially high energy use, facilities and equipment needs, intensive water treatment, together with some species-specific volume and flow requirements reduce land-based systems application to certain species, life stages, and locations. Significant treatment may be necessary to make incoming water suitable as a culture medium, as well as waste-water treatment to prevent chemical (although seaweeds are well-suited at minimizing nutrient load before effluent release) and biological pollution, prevent transmission of diseases, and prevent spread of introduced species from effluent (Matson et al., 2006; Hafting et al., 2012; Pereira et al., 2024). Monitoring may be necessary to ensure and document success of those goals, and regulatory compliance.

Although many species of red, green, and brown macroalgae have successfully been grown in land-based systems (Evans and Langdon, 2000; Demetropoulos and Langdon, 2004; Langdon et al., 2004; Pang et al., 2009; Hamilton et al., 2022; Moreira et al., 2022), both in research and commercial production (given advantages of their particularly plastic morphology), kelp species are comparatively rare in land-based culture and may present specific challenges. However, there are reports of successful tumble culture of cold water kelp species in the United States (Redmond et al., 2014) and potential for tank culture of kelp species (Purcell-Meyerink et al., 2021).

Conclusion

Kelp farming has generated considerable interest in recent years, both locally and nationally, sparking the need for comprehensive evaluations of the industry's impact on local ecosystems. Here, we provide an initial review of potential kelp farming impacts for the Salish Sea, and recommendations for how to begin this commercial practice so as to maximize those benefits and minimize potential harms. Due to a lack of regionally-specific information, we looked broadly at kelp farming practices across the world to inform our recommendations, in addition to considering some socio-ecological issues unique to the Salish Sea.

We found that in general, kelp farming poses few risks to the natural environment, and indeed, may provide many benefits to the surrounding system. However, we recommend careful consideration of native eelgrass beds and other marine vegetation, whale migration and feeding areas, forage fish spawning areas, and native stock availability for sourcing seed when developing farming projects. We also note that following outlined recommendations would not necessarily ensure receiving local, state, and federal authorizations for particular farms. To this end, spatial planning tools, and analyzes similar to Aquaculture Opportunity Areas in Southern California and the Gulf of Mexico (NOAA, 2020), are needed to help guide regulators and potential farmers in determining ideal site locations. In addition to ecological considerations, it is critical that potential farmers coordinate or partner with tribes early on to reduce any potential conflict with usual and accustomed area fisheries (key considerations are summarized in Figure 2). We recommend prioritizing siting farms in areas with pre-existing infrastructure or with low fisheries activity to minimize potential conflicts. With these considerations in mind, we believe that kelp farming may have a promising future in the Salish Sea but will require monitoring and an adaptive management approach to foster social acceptance and economic viability.

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Characterization of the U.S. Eastern Gulf of Mexico Reef Fish Bottom Longline Catch through Fishery Collaboration with Electronic Monitoring

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Introduction

The U.S. Gulf of Mexico (hereafter referred to as "Gulf") commercial reef fish fishery involves more than 800 federally permitted vessels (Stephen[1]) using primarily bottom longline (BLL) and vertical hook-and-line (VL) gear (Scott-Denton et al., 2011; Pulver and Stephen, 2019). In the Gulf, off Florida's west coast, about 62 vessels have an eastern Gulf reef fish BLL endorsement (Stephen[1]). These vessels primarily target red grouper, *Epinephelus morio*, in shallow waters, and yellowedge grouper, *Epinephelus flavolimbatus*; tilefish (Malacanthidae); and snappers, *Lutjanus* spp., in deeper water (Scott-Denton et al., 2011).

Thirty-one of the reef fish within this multi-species complex in federal waters are managed by the Gulf of Mexico Fishery Management Council (GMFMC[2]) who implement plans or regulations based on quantitative scientific information provided through fish stock assessments to determine stock status and set annual catch limits to prevent overfishing (Mace et al., 2001). The Gulf reef fish industry and management agencies have an increased urgency to provide reliable data with the Magnuson-Stevens Fishery Conservation and Management Reauthorization Act of 2006[3] requiring implementation of annual catch limits (ACL's) and accountability measures (AM's) to curtail overfishing (Farmer et al., 2016). Unfortunately, for many of the reef fish stocks there is limited data available for input into stock assessments (Farmer et al., 2016) and to inform management actions in support of sustainable fisheries (Lynch et al., 2018). Resulting uncertainty in a stock's status makes it difficult to be confident that overfishing is not occurring and the stock is not overfished (Methot, 2015). Ultimately, robust, timely, and accurate catch, effort, and discard data is needed to support species-specific stock assessments and science-based management (Michelin et al., 2018).

The Gulf commercial reef fish fishery has relied on multiple sources for fishery-dependent data contributions, including onboard observers, dealer reports, port sampling, trip interviews, vessel monitoring systems (VMS), and captains' self-reported logbooks, all of which have their own data limitations. The National Marine Fisheries Service (NMFS) reef fish observer program, an important source for fishery-dependent data, is limited to sampling approximately 2% of the commercial reef fish trips annually, due to bud-

[1]Stephen, J., Ph.D., personal commun., 1 April 2022. LAPP/DM Branch Chief, NOAA, NMFS Southeast Regional Office, 263 13th Ave South, St. Petersburg, FL 33701.

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[2]GMFMC (Gulf of Mexico Fisheries Management Council. NOAA, Natl. Mar. Fish. Serv. Federally Managed Gulf of Mexico Reef Fish. (https://www.fisheries.noaa.gov/species/federally-managed-gulf-mexico-reef-fish).

[3]Magnuson-Stevens Fishery Conservation and Management Reauthorization Act of 2006 (https://www.govinfo.gov/content/pkg/CHRG-112shrg67219/html/CHRG-112shrg67219.htm).

ABSTRACT— Integration of electronic monitoring on commercial bottom longline vessels has been successfully conducted through several pilot studies in the U.S. Gulf of Mexico reef fish fishery. Collaborative work conducted from 2016 through 2021 by the Mote Marine Laboratory Center for Fisheries Electronic Monitoring at Mote (CFEMM) and 13 volunteer eastern Gulf fishing vessels resulted in observations of 82,936 individual fish from 131 unique species or species groups from 306 trips covering 2,822 sea days of fishing effort. The fishing area covered 114,268 km² with capture depths ranging from 35 m to 360 m. The most predominantly caught species were red grouper, Epinephelus morio *(65%); red snapper,* Lutjanus campechanus *(9%); and yellowedge grouper,* Epinephelus flavolimbatus *(5%). Of these species, 48%, 23%, and 2%, were discarded, respectively. Important bycatch species included blueline tilefish,* Caulolatilus microps*; scamp,* Mycteroperca phenax*; and gag grouper,* Mycteroperca microlepis*. Notably, sea turtle and sea bird interactions were minimal. Discards included undersized and depredated reef fish, along with a variety of shark species. Statistically significant spatial clusters of high catch per unit of effort (CPUE) hotspots, low CPUE cold spots, and random distributions are presented spatially joined to 10.0 min grids for primary targeted and bycatch species, grouped shark bycatch, and specifically for sandbar sharks,* Carcharhinus plumbeus*, a species that is prohibited from harvest, except under a NMFS research permit. This project demonstrates that electronic monitoring is an effective method for collecting catch and bycatch data from the commercial eastern Gulf of Mexico reef fish fishery. Increasing trip coverage and providing additional EM-derived data that complements observer reports would contribute to enriching the data available for supporting stock assessments and promoting sustainable fishing practices.*

get constraints (Scott-Denton[4]). While data is plentiful on landed commercial catches (Stephen and Harris, 2010), these data do not reflect all of the fish that were caught, as discards can represent a large proportion of the total catch (van Helmond et al., 2014). This can be attributed to the fishery's high species diversity in combination with multiple fishing gear types and methods, which often results in the incidental captures of non-target (bycatch) or undersized species (Pulver and Stephen, 2019). Of fundamental concern to fishery managers is the contribution of discards to the overexploitation of stocks (Sissenwine et al., 2014, as cited in Pulver and Stephen, 2019). The fishermen have the opportunity to provide bycatch and corresponding discard information in voluntary self-reported discard logbooks, but historically these have been incomplete or questionable with reports of trips with "no discards," which is highly unlikely in a multispecies fishery where size and harvest limits necessitate regulatory discards (GMFMC[5]). Pulver and Stephen (2019) reported that biases associated with the commercial fishermen logbooks primarily result from inaccurate reporting of species that are caught in large numbers, are of little economic interest, and/or are from low compliance rates.

To supplement the collection of fishery-dependent data, electronic monitoring (EM) video technology has gained prominence over the last two decades as an additional approach for documenting catches in fisheries (Ames et al., 2007; Michelin et al., 2018; Bradley et al., 2019; Emery et al., 2018; van Helmond et al., 2020; Fujita et al.[6]). These systems typically include video cameras, a global positioning system (GPS), and gear sensors to record fishing activity. Similar to at-sea observers, EM offers advantages over gathering data on a particular fishery solely through monitoring portside landings (Ames et al., 2007), as it enables observations of fishing location coordinates, catch location and quantity, composition of the catch, and discarded fish. The permanent documentation of this information through video and sensors for subsequent review contributes to reducing data uncertainty (Ames et al., 2007; Michelin et al., 2018). Numerous studies have concluded that EM technology whether designed to augment onboard human observer programs or serve as a stand-alone application can effectively function as a data collection platform to broaden monitoring coverage of fishing activities (McElderry et al., 2003; Ames et al., 2007; Michelin et al., 2018; Bradley et al., 2019; Gilman et al., 2019; Wozniak et al., 2020). These findings have increasingly led to the integration of EM technology on commercial fishing vessels across various global fisheries, particularly in those where human observer coverage did not exist or where it is limited (Michelin et al., 2018; Bradley et al., 2019; Gilman et al., 2019; van Helmond et al., 2020).

In the Gulf region, EM systems have proven effective on commercial reef fish vessels in several pilot studies. These studies, notably those first conducted by Archipelago Marine Research (AMR), Ltd., Victoria, Canada, demonstrated the reliability of EM in documenting fishing effort and retained catch on BLL vessels (Pria et al.[7]). However, some limitations were noted that EM was less reliable for determining catch discards or species identification, which were mostly attributed to camera obstructions or dirty lenses (Pria et al.[7]; NOAA[8]). During a subsequent study in 2012–13, a collaborative effort led by the Ocean Conservancy, St. Petersburg, FL, with partners AMR, Gulf of Mexico Shareholders' Alliance, Galveston, TX, and Mote Marine Laboratory (Mote), Sarasota, FL, engaged both commercial BLL and VL vessels to further assess EM's suitability for capturing fishery data and validating captains' self-reported data. This initiative, eventually led Mote to spearhead the expansion of EM with a focus on the eastern Gulf starting in 2014. This expansion involved collaborations with various organizations, including AMR, Waterinterface Ltd., Wimauma, FL; Sustainable Fisheries Partnership Foundation, San Francisco, CA; Environmental Defense Fund, New York, NY; and the Gulf of Mexico Reef Fish Shareholders' Alliance, Galveston, TX. The expansion efforts not only included the addition of vessels, but also aimed to effectively address the limitations observed in the previous pilot studies. This encompassed improvements in camera coverage, refinement of documentation processes, addition of species for selection, and categorization of catch, non-targeted species of bycatch, and discards.

In 2016, Mote initiated a significant change by transitioning to a new EM hardware and software provider, Saltwater Inc., based in Anchorage, AK. This transition was driven by a National Marine Fisheries Service request for Mote to incorporate non-proprietary review and vessel software. The shift of adopting the non-proprietary EM vessel and review software brought several advantages, including reduced software costs and increased user flexibility, transparency, and accessibility.

[4]Scott-Denton, E., Ph.D., personal commun. 16 June 2020. Fishery Observer Program Manager, NOAA Southeast Fishery Science Center, Galveston Laboratory, 4700 Avenue U, Galveston, TX 77551, USA.

[5]Gulf of Mexico Fishery Management Council (GMFMC), Sustainable Fisheries Committee. January 26, 2022. (Webinar Transcript) (https://gulfcouncil.org/wp-content/uploads/E-2-GMFMC-Sustainable-Fisheries-Minutes-January-2022.pdf).

[6]Fujita, R., C. Cusack, R. Karasik, H. Takade-Heumacher, and C. Baker. 2018. Technologies for improving fisheries monitoring. Environmental Defense Fund, San Francisco. 71. (https://fisherysolutionscenter.edf.org/sites/default/files/Technology_for_Improving_Fisheries_Monitoring.pdf).

[7]Pria, M. J., H. McElderry, M. Dyas, and P. Wesley. Using electronic monitoring to estimate reef fish catch on bottom longline vessels in the Gulf of Mexico: a pilot study. Unpublished report prepared for the National Marine Fisheries Service by Archipelago Marine Research Ltd., Victoria, British Columbia, Canada, 42 p. (https://www.yumpu.com/en/document/read/11507510/using-electronic-monitoring-to-estimate-reef-fish-catch).

[8]National Oceanic and Atmospheric Administration (NOAA). 2015. Electronic monitoring and reporting implementation plan - Southeast Region. January 8. (https://gulfcouncil.org/wp-content/uploads/E%20-%205%20EM%20ER%20Implementation%20Plan.pdf).

In addition to the non-proprietary software, Saltwater Inc. provided a more compact EM processor system that was particularly useful for vessels with limited space. This, coupled with the availability of in-state technical service staff, improved convenience, and lowered costs for installations and support for vessels utilizing the system.

In 2020, the Center for Fisheries Electronic Monitoring at Mote (CFEMM) was established with the goal of further enhancing fishery-dependent monitoring in the reef fish fishery utilizing EM as a tool. This initiative sought to augment traditional observer reporting by providing a complementary and supplementary method for data collection. The overarching objective of the CFEMM was to provide EM data products to inform industry stakeholders and Gulf management, assisting them in their efforts to enhance fisheries management practices.

This paper aims to demonstrate that EM technology applied in the eastern Gulf commercial reef fish BLL fishery is a valuable tool, capable of providing accurate, detailed characteristics of catch, bycatch, and discards. It highlights the significant contributions of volunteer industry stakeholders in generating a substantial amount of EM data on fishing operational metrics. Through general additive models, it examines temporal catch patterns and the spatial distribution of primary target and prominent bycatch species through hotspot/coldspot analysis. Additionally, the paper reports incidental sea turtle and sea bird catches, along with instances of catch depredation (damage) by sharks and/or marine mammals, documented by EM from July 2016 through December 2021. These findings contribute to a more comprehensive understanding of the fishery's spatial dynamics and serves to inform management for their consideration in efforts aimed to promote sustainable fishing practices for the long-term health of this fishery.

Methods

Through established industry partnerships, 13 BLL vessels from Madeira Beach, Redington Shores, and Cortez, Florida, voluntarily participated in pilot projects requiring the onboard installation of an EM system. Saltwater Inc. EM systems included an electronic monitoring unit (EMU) consisting of a Lanner control center computer processor (Model 5770-7D; Lanner Electronics, Inc, Fremont, CA) housed in a protective weather resistant and non-jarring case mount. The processor held two exchangeable 1-terabyte data hard drives, and stored data was encrypted through the EMU Linux operating system (OS). Each vessel was equipped with three or four closed-circuit, fixed-focal-length digital Internet Protocol (IP) cameras with light emitting diodes of one type or a combination of GeoVision Inc., Taiwan (Models GV-EVD3100 [3MP H.264, 1/2.8" progressive scan, low lux], and/or (GV-TDR2700 [2MP, 2.8, H.265, 2.8mm, low lux]), and or Vivotek Inc., Taiwan (Model FD8134V [1MP, H.264 IP66]).

Cameras were mounted using custom stainless steel brackets or a camera base plate based on captain and crew input; initial onboard testing confirmed that camera locations provided optimum coverage of fishing operations, including where catch was processed or discarded. Most vessels were also installed with custom aluminum retractable or stationary roof mounted vertical or horizontal booms to position one or more cameras overhead and/or extended over the rail to improve coverage of haul and discard areas. A multi-port power over Ethernet (POE) switch connected marine Category 5 Enhanced (CAT5e) camera cables to the EMU. An inline hydraulic sensor and a set of two magnetic reverse polarity rotation sensors installed on a mainline drum served to document when the vessel was setting and hauling gear and triggered video recording of gear retrievals. Video recording with no sound was activated at the start of the mainline being hauled and continued to approximately 30 min post-haul, based on software settings to allow for complete coverage of fish processing. A rooftop or cabin window mounted GPS receiver logged the vessel position, speed, and heading. The sensors and GPS recorded data continuously when the processor was powered on. A wheelhouse monitor displaying all camera views was used by the captain to watch deck activities in real time and for confirming EM system functionality.

Captains and crew were requested to power on their EMU prior to leaving port and to keep the system on during their fishing efforts. Systems could be powered down when the vessel wasn't fishing or if there were concerns of power limitations during their night fishing activities. Upon trip completion, hard drives were retrieved and replaced with reformatted hard drives. During vessel hard drive collections, captains and crew members provided additional trip and or system performance information.

At the CFEMM, vessel data was downloaded from the hard drives using an Intel Next Unit of Computing (NUC) processor with Saltwater Inc. software for conversion of the Linux encrypted data to a more user-friendly Microsoft Windows OS format. The resulting imagery and sensor data was subsequently stored on a dedicated server for retrieval to individual computer stations for review and backed up both pre- and post-review on a series of secure Synology Networked Attached System (NAS) units.

Imagery review was conducted using Saltwater Inc. non-proprietary review software modified specifically for this fishery. A preliminary review of the recorded sensor and video footage was made to identify which set-haul events (SHE's) were acceptable for full review and analysis. If a SHE was deemed not reviewable due to incomplete imagery or sensor information, it was removed from the individual trip dataset, reducing the total selectable events. Only reviewable SHE's remained in the trip data pool, from which 25% were randomly sampled from each trip using a "random-sampling-without-replacement" method.

Data associated with fishing activities represented trip, SHE, and catch-

level information. Trip details recorded by the system included the trip duration, time, and location of sets and hauls. In the review software the video reviewers selections were keyed into a custom template from a series of dropdown menus. Trip level entries included the offload port, whether an observer was onboard, and specific gear characteristics, including longline leader material and hook type. Confirmation of hook sizes used were documented during vessel visits, and provided captain questionnaires. Bait types were also selected at the overall trip level based on reviewed hauls. Information collected at the SHE level included the timing and location of the gear set and haul. Each caught fish was identified to the lowest taxonomic level possible, with more than 200 individual species or species grouping options available for selection. An "unidentified" code was available when identification could not be discerned. For sharks that could not confidently be identified to species level, particularly those released at the rail while underwater, three classification groupings were available for selection including, shark unidentified, carcharhinid unidentified, and hammerhead unidentified. Individual fish catch characteristics, including handling, condition upon capture, and fate selection options included 1) Handling: brought onboard, not handled (dropped off), cutoff at rail (no entanglement), cutoff at rail (entanglement), or unknown handling; 2) Condition: live healthy, live stomach and/or eyes protruding, live damaged, dead (damaged), dead (undamaged), or unknown condition; and 3) Fate: retained, retained as bait, discarded live healthy (vented), discarded live healthy (not vented), discarded live damaged (vented), discarded live damaged (not vented), discarded dead, discarded unknown, or unknown fate. Specific to shark bycatch, juvenile or adult status was selected and an estimated size category of small (< 1 m), medium (1 to 2 m), or large (> 2 m) was assigned. If possible, sex was recorded based on the presence or absence of claspers. Unknown maturity and sex were recorded when neither could be determined.

A series of manual and automated quality control steps were taken with the resulting species detailed recorded information, referred to as annotations, prior to export into a Microsoft Access database, including confirmation of species identification, accuracy of assigned condition, and fate. Trip dataset annotations were aggregated using an R statistical software routine[9], which enacted 75 explicit error checks. Aggregated data underwent thorough checks for missing information and inconsistencies, such as handling codes that did not align with fish fate. Any identified issues were automatically detailed in a report, allowing the opportunity to resolve any problems prior to importing the dataset into the final database. Finalized data were then spatially joined using R code to link metadata including bathymetry information to enable depth estimation for catch events.

For the presented work, effort was calculated as the product of hook number and soak time in hook-hours. Soak time was defined as the start of the longline (mainline) deployment through the recovery of the last hook (McCarthy[10]). The standard regulatory hook count for this fishery was 750 hooks per longline (NMFS[11]). This number was used as the estimated amount of hooks utilized per set to calculate effort. Catch rates were calculated by number of fish (both species-specific and total number) per hook-hour scaled by 1,000 hooks to standardize the CPUE calculations, following a protocol used by Scott-Denton et al. (2011).

[9]R Core Team. 2020. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria (https://www.R-project.org/)

[10]McCarthy, K. J., Ph.D., personal commun. 2017 April 11. Branch Chief, NOAA Southeast Fisheries Science Center, 75 Virginia Beach Drive, Miami, FL 33149.

[11]National Marine Fisheries Service (NMFS). Final rule modifies the number of unrigged hooks carried on board bottom longline vessels in the Gulf of Mexico (https://www.fisheries.noaa.gov/bulletin/final-rule-modifies-number-unrigged-hooks-carried-board-bottom-longline-vessels-gulf).

Generalized Additive Model (GAM) Methods

Inter-annual patterns in CPUE estimates using GAM's were depicted (Hastie and Tibshirani, 1987). For each species of interest, the GAM applied a nonparametric smoothing function for data and was fitted using the GAM function in R[9]. The degrees of freedom associated with fitting each smoothing function (k) were chosen by identifying local minima in generalized cross validation scores; the smallest k value associated with a local minima was selected for each GAM.

Geographic Information System (GIS) Methods and Spatial Analysis

Spatial analysis was based on a stepwise procedure using geospatial statistics that leveraged spatial heterogeneity inherent in fishing activities. The uniquely precise locations of individual fish annotations provided by EM systems functioned as proxies for catch location and covariate responses for assessment of spatial distributions including global spatial autocorrelation (Global Moran's I), local indicators of spatial association (LISA's), and response variable density maps. Therefore, areas of concentrated fishing effort and fine-scale aggregations of target and non-target fishes were identified and used to describe particular areas of fishing activity in terms of localized fish density.

Fishing activity was quantified by use of a convex polygon to define the fishing area encompassing all annotated catch, and kernel density was used to illustrate fishing effort intensity. Fishing effort for all species, all vessels, and all years reported was defined and mapped using simple point density analysis, with raster cell size and search radius determined following the methods of Scott-Denton, et al. (2011), executed in ArcGIS Pro 2.8[12]. To derive point density rasters, the analysis em-

[12]ESRI. 2020. ArcGIS Pro Desktop. Release 2.8. Environmental Systems Research Institute, 380 New York Street, Redlands, CA. 92373 (https://www.esri.com/en-us/arcgis/products/arcgis-pro/overview).

ployed a search radius of 25 km and an isotropic circular cell size of 5,000 m, with results reported as hook-hours/km^2. The Jenks Natural Breaks Classification (Jenks, 1967) was used for raster maps to minimize the average deviation from the class mean while maximizing the deviation from the means of all other groups. Annotated catch and initial set locations were mapped using the World Geographic Coordinate System GCS_WGS_1984. Specific locations and distances were mathematically projected using a transformation from spherical coordinates (latitude and longitude) to an XY (planar coordinate system) known as WGS_1984_World_Mercator, WKID-3395 Authority-EPSG. Additionally, LISA's, referenced as "hot spot" and "cold spot" analyses, were primarily analyzed using the local spatial statistic Optimized Hot Spot Analysis, executed in ArcGIS Pro 2.8[12] following Getis and Ord (1992) and Ord and Getis (1995) for species-specific CPUE, all sharks CPUE, and red grouper discard proportion. False positives were evaluated by the statistic during execution and false discovery executed if optimal. Z-score, p-value, and confidence level bin (Gi_Bin) were the final results of the analysis. Features in the ±3 bins reflect statistical significance with a 99% confidence level; features in the ±2 bins reflect a 95% confidence level; features in the ±1 bins reflect a 90% confidence level; and the clustering for features in bin 0 were not statistically significant. A 0.05 significance level was selected for spatial hot spot identification. Hot spot and cold spot centroids were averaged, aggregated, and spatially joined to 10 min grid cells (10.0 min x 10.0 min; 18.5 km x 20.4 km; 377.4 km^2) as areal features using a one to one operation; match option = within; merge rule = mean, final CPUE Gi bins were aggregated to grids as mean bin probability and were symbolized accordingly.

Results

Vessel Participation and Sampling Effort

From July 2016 through December 2021, 13 eastern Gulf commercial BLL reef fish vessels provided EM video from 306 trips covering 2,822 sea days. Most vessels fished for periods of 10–14 days with a crew of 2–4. Vessel lengths ranged from 37.2–48.4 ft (11.34–14.75 m) and each used a single drum and one mainline made of either braided stainless steel or monofilament averaging 4.95 nmi (9.2 km). Monofilament gangions (250–300 lb test) of 3–6 ft (1–2 m) in length were set with up to 750 baited non-stainless steel circle hooks as per NMFS regulations (GMFMC[13]). The participating reef fish vessels generally used Mustad, Miami, FL, offset circle hooks from a size of 12/0 to a larger size of 15/0, with 13/0 being the most commonly used option. The frequency of longline deployments ranged from two to four per day, with most occurring during daylight hours. Typical soak times were 3–4 h, from the beginning of the set to the end of the haul.

The area of fishing activity encompassed 114,268 km^2 and spanned the northern Gulf from near the Madison Swanson marine protected area to south, proximate to the Dry Tortugas. This region reflects the U.S. Gulf statistical zones 2–6 and 8–9, with depths ranging from 34 m to 360 m at the edge of the Florida Escarpment (Fig. 1). The cumulative fishing effort amounted to 5,159,933 hook-hours. Predominantly, the fishing effort occurred in zones 4 and 5, constituting 31% (1,584,151 hook-hours) and 38% (1,939,577 hook-hours) of the total effort, respectively. Notably, these zones represent only 39% of the total fishing area defined by the fishing polygon in Figure 1.

In zones 4 and 5, fishing effort was generally located proximal to the vessel's home ports, centered in three high density areas, including lat. 27°30'N long. 83°30'W, lat. 26°30'N long. 83°20'W, and lat. 27°0'W long. 83°20'N. Highest density fishing effort kernels of approximately 16,000–19,000 hook-hours/km^2, encompassed large areas in zone 5, approximately 149.0 km^2 and 413.6 km^2, respectively (Fig. 1). In zone 4 the fishing effort covered approximately 43.8 km^2. No fishing events were recorded in zone 7.

Fishery-dependent data presented here resulted from the review of 1,796 hauls representing the established 25% of all potentially analyzable SHE's. The remaining 75% of SHE's were archived. Annual video sampling rates are shown in Figure 2. Subsequent event review resulted in 82,936 individual annotated fish (Fig. 3) from 131 unique species or species groups.

Bait

Bait types used during the recorded trips, though not necessarily each haul, mirrored the typical selection used in this fishery during the specified time period. The choice of bait varied based on a captain's preference, availability, and opportunities presented by bycatch. On most trips there was a mix of different baits used, with herring being the most common, utilized in 65% of the trips. Eels, remoras, and grouper stomachs made up the majority of the caught bait category, accounting for 63% of trips. Assorted fish scraps, known as trim, were presented on 21% of trips and included scraps such as bloodlines brought on the trip from fish processing, consisting of a variety of fresh and saltwater species. Squid was used on 14% of trips, exhibiting a decrease over time compared to earlier periods. Salted shark belly meat acquired from the commercial shark fishery was used on 4% of the trips.

Catch Composition and Characterization

All BLL catches recorded through EM are presented in Table 1, with the most predominant species being red grouper (n = 53,740), constituting 65% of the total recorded catch. The average capture depth for this species was 52 m, with minimum and maximum depths of 36 m and 130 m, respectively. Red grouper were recorded with physical attributes of barotrauma 42% of the time (Table 2), and fish-

[13]Gulf of Mexico Fishery Management Council (GMFMC). 2019. Commercial fishing regulation for Gulf of Mexico federal waters. Tampa, FL, 52 p. (https://gulfcouncil.org/wp-content/uploads/commercial-regulations.pdf).

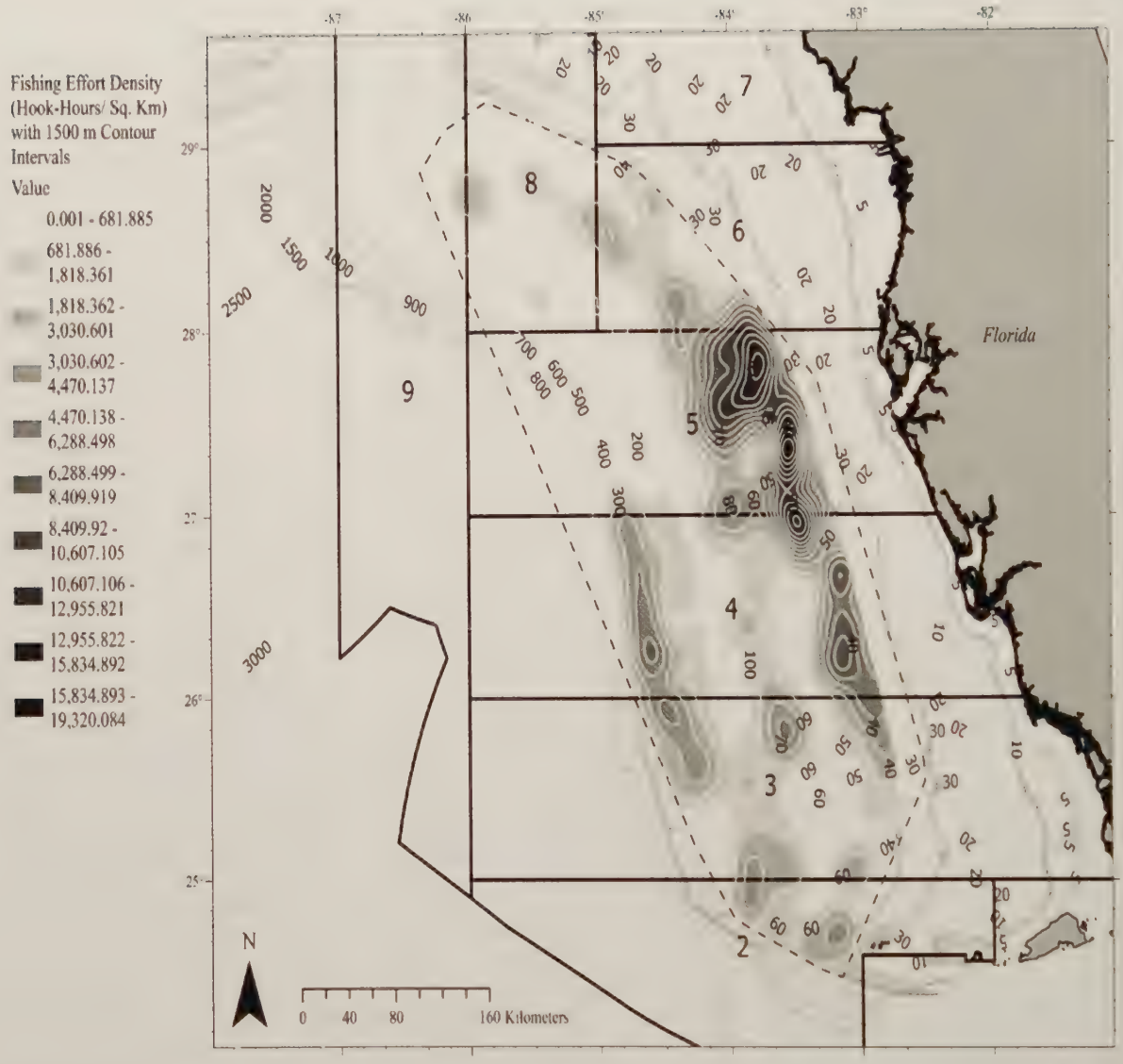

Figure 1.—Fishing area (dashed line convex polygon) with kernel density raster partitioned by 1,500 m contours overlaying NMFS statistical zones 2-9 to define participating electronic monitoring vessels fishing areas and kernels of most intense fishing effort with bathymetry from July 2016 through December 2021.

ermen were twice as likely to vent the fish prior to release than not (Table 3). This primary target species was documented as damaged by predators, evidenced by a partial carcass or distinct rake marks, at a rate of 1%. Nearly half of red grouper (48%) were discarded, although rates varied across the fishing area with higher discard rates in shallower depths shoreward of 50 m (Fig. 4). The largest aggregations of high confidence (≥ 95%) CPUE hot spots occurred in 19 grid cells, centered at lat. 27°57'N, long. 83°73'W, comprising an area of 7,163 km^2 (Fig. 5). This area of highly efficient fishing also approximates the same general locations where fishing effort was the highest (Fig. 1). High confidence CPUE hot spots included an additional 17 grid cells from approximately lat. 26°93'N, long. 83°57'W to lat. 25°60'N, long. 82°74'W, running northwest to southeast, which generally approximated the 50 m isobath, and comprised approximately 6,416 km^2. An additional grouping of three cells was also pres-

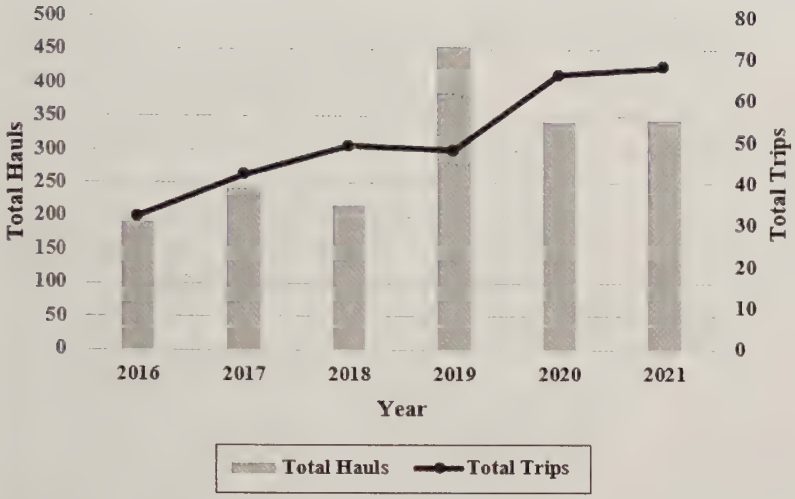

Figure 2.—Total hauls reviewed and total trips recorded from participating eastern Gulf of Mexico bottom longline vessels.

ent in the northern most area adjacent to the Florida Middle Grounds at lat. 28°45'N, long. 84°20'W. Red grouper CPUE was generally low (cold spots) seaward of the 60 m isobath. Peaks of CPUE for this species occurred annually between October and March throughout the study period (Fig. 6), straddling the June through September BLL closure inside 35 fathoms interannually (Fig. 7).

Similar observations were seen for red snapper (Fig. 6), which made up 9% of the catch (n = 7,154). Approximately one quarter were discarded and venting was recorded during half of the discard events. Predation was recorded for 2% of the individuals (Table 3). Capture depth ranged from 36 m to 168 m with an average of 64 m. The CPUE hot spots were located in two main groupings (Fig. 8). The largest aggregation of high confidence CPUE hot spots occurred in nine grid cells centered at lat. 27°66'N, long. 84°06'W and comprised an area of 3,397 km^2. A northern group centered at lat. 28°53'N, long. 84°86'W comprised five grid cells and an area of 1,887 km^2. There were two additional red snapper hot spot grid cells located in the south central and southern part of the fishing area, centered at lat. 29°17'N, long. 88°62'W and lat. 24°85'N, long. 83°26'W, respectively. A conspicuous area of red snapper cold spots was located inshore between 30 and 40 m isobaths centered at lat. 27°50'N, long. 83°54'W and located in an area of intense red grouper fishing with high catch efficiency. The CPUE rates for this species were relatively stable, with the exception of elevated CPUE in 2018 (Fig. 6).

Yellowedge grouper was the most targeted reef fish species on trips that focused on the deepwater grouper and tilefish complex, resulting in the third highest number of annotations (n = 4,237; 5% of the catch). Average capture depth was 213 m, with the shallowest at 69 m, and the deepest from 301 m. Due to a lack of size limits, total discards for this species were minimal (2%), with most of the discards attributed specifically to observed physical predator damage (1%), or juvenile individuals. The CPUE hot spots were aggregated in grid cells approximating the 100–300 m isobaths in northern and southern groups centered near lat. 28°84'N, long. 85°73'W and lat. 26°21'N, long. 84°21'W, respectively (Fig. 9). There were 6 complete high confidence grid cells and 2 partial cells in the northern group while the southern grouping contained 12 cells and 1 partial cell. Widespread cold spots were found adjacent to these hotspot groupings. The CPUE for this species was highest in 2017 (Fig. 6), and during times of the year where red grouper were targeted less (Fig. 7).

Blueline tilefish, *Caulolatilus microps*, were most often caught (n = 3,077) while fishermen targeted yellowedge grouper in similar average depths of 213 m. Despite the absence of a regulatory size limit for this species, the reluctance of the fishermen to utilize quota for retaining them led to high grading. This involved discarding them (17%) or using them as bait (5%), allowing for the retention of species with higher market value. Their CPUE hot spots were mainly aggregated between the 100–300 m isobaths in two distinct groupings centered near lat. 26°70'N, long. 84°46'W (three grid cells) and at lat. 25°92'N, long. 84°47'W with two partial and one full grid cell (Fig. 10). The CPUE for this species generally increased over the period and was the highest in late 2020 (Fig. 6).

Scamp, *Mycteroperca phenax*, catches ranged in depths from 44 m to 178 m with 1,127 individuals recorded. Discards were minimal (3%) with few undersized or damaged individuals. Their hotspots were typically further offshore than those of red grouper, with an average depth of capture of 83 m. A large northern group of six high confidence cells was centered near lat. 27°91'N, long. 84°44'W, comprising an area of 2,264 km^2. A single hot spot cell occurred along the same isobath near lat. 27°26'N, long. 84°06'W, and a rather deep partial hotspot cell was located at the 200 m isobath near lat. 25°44'N, long. 84°39'W (Fig. 11). The CPUE for this species was nearly twice as high when this study was initiated in late 2016 than in following years, though relatively elevated CPUE was seen during spring 2018 and fall 2021 (Fig. 12).

Similarly, gag grouper, *Mycteroperca microlepis*, capture (n = 960) depth ranged from 37 m to 163 m with an average depth of 78 m. Overall discards were also low (8%) when compared to red grouper and red snapper, as under-

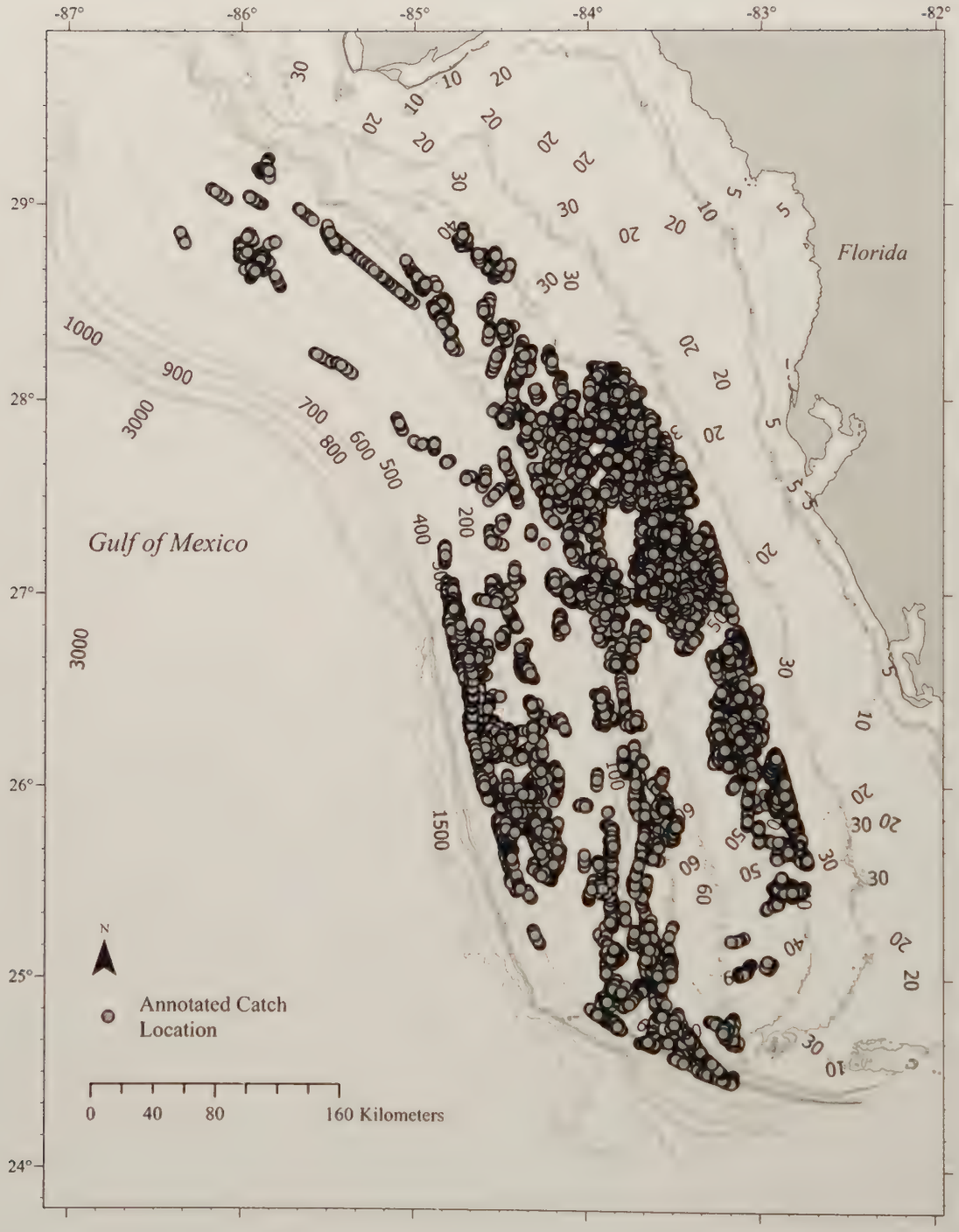

Figure 3.—Annotated fish (n = 82,936) catch locations in the eastern Gulf of Mexico.

sized individuals were infrequently encountered. Discounting individuals of this species that were lost at the surface (broke leader or fish dropped off) and those with predator damage, gag grouper generated a discard rate of 2%. Their catch rates did not indicate significant clustering between 28°N and 24°N. A single group of 12 adjacent cells (Fig. 13) were located around a center point at lat. 28°57'N, long. 84°91'W, covering 4,529 km^2. The CPUE for gag grouper nearly mirrored trends seen in scamp during 2017–21, with the exception of comparatively lower CPUE in 2016 (Fig. 12).

Shark Bycatch

Twenty species of sharks were caught (Table 1), totaling 5% of all annotated catch (n = 3,973). When comparing sharks separately from the rest of the catch, the Atlantic sharpnose shark, *Rhizoprionodon terraenovae* (23%), and sandbar shark, *Carcharhinus plumbeus* (15%), were the most commonly caught, followed by the blacknose shark, *Carcharhinus acronotus* (13%), smooth dogfish, *Mustelus* spp. (11%), and spiny dogfish, *Squalidae* (9%). Smooth dogfish (n = 428) were grouped, potentially representing three species of *Mustelus* present in the region (Giresi et al., 2015). The spiny dogfish category (n = 378) consisted of a variety of deepwater dogfish species, including Cuban, *Squalus cubensis*; shortspine, *Squalus mitsukurii*; and roughskin, *Cirrhigaleus asper*, but primarily consisted of the smaller Cuban and shortspine dogfish. The larger roughskin dogfish were rarely encountered. Cuban and shortspine dogfish in particular were difficult for reviewers to differentiate and recent discoveries of new dogfish species in the region complicated identification (Pfleger et al., 2018). Of the sharks recorded, 18% were species that were prohibited to harvest both commercially and recreationally in state and federal waters. These included sandbar, night, *Carcharhinus signatus*; dusky, *C. obscurus*; sixgill, *Hexanchus* spp.; sharpnose sevengill, *Heptranchias perlo*; and angel sharks, *Squatina dumeril*.

Table 1.—Species in descending frequency of occurrence for participating bottom longline vessels fishing the eastern Gulf of Mexico from July 2016 through December 2021.

Common name	Scientific name	Number caught	Relative frequency (%)
Red Grouper	*Epinephelus morio*	53,740	64.80
Red Snapper	*Lutjanus campechanus*	7,154	8.63
Yellowedge Grouper	*Epinephelus flavolimbatus*	4,237	5.11
Blueline Tilefish	*Caulolatilus microps*	3,077	3.71
Scamp	*Mycteroperca phenax*	1,127	1.36
Gag Grouper	*Mycteroperca microlepis*	960	1.16
Atlantic Sharpnose Shark	*Rhizoprionodon terraenovae*	912	1.10
Mutton Snapper	*Lutjanus analis*	776	0.94
Snowy Grouper	*Epinephelus niveatus*	683	0.82
Jolthead Porgy	*Calamus bajonada*	612	0.74
Sandbar Shark	*Carcharhinus plumbeus*	597	0.72
Blacknose Shark	*Carcharhinus acronotus*	514	0.62
Tilefish, Golden	*Lopholatilus chamaelonticeps*	459	0.55
Speckled Hind	*Epinephelus drummondhayi*	430	0.52
Dogfish, Smooth	*Mustelus* spp.	428	0.52
Moray Eel, Unidentified	*Muraenidae*	419	0.51
Lane Snapper	*Lutjanus synagris*	406	0.49
Dogfish, Spiny	*Squalidae*	378	0.46
Eel, Unidentified	*Anguilliformes*	375	0.45
Red Porgy	*Pagrus pagrus*	345	0.42
Gray Snapper	*Lutjanus griseus*	325	0.39
Tiger Shark	*Galeocerdo cuvier*	316	0.38
Remora	*Echeneidae*	307	0.37
Great Barracuda	*Sphyraena barracuda*	271	0.33
Nurse Shark	*Ginglymostoma cirratum*	237	0.29
Almaco Jack	*Seriola rivoliana*	227	0.27
Vermilion Snapper	*Rhomboplities aurorubens*	206	0.25
Pufferfish	*Tetraodontidae*	202	0.24
Gray Triggerfish	*Balistes capriscus*	194	0.23
Greater Amberjack	*Seriola dumerili*	190	0.23
Lizardfish	*Synodus* spp.	183	0.22
Shark, Unidentified	*Selachimorpha*	183	0.22
Hake, Unidentified	*Gadiformes*	176	0.21
Blackfin Tuna	*Thunnus atlanticus*	165	0.20
Silky Shark	*Carcharhinus falciformis*	148	0.18
Little Tunny	*Euthynnus alletteratus*	146	0.18
Toadfish	*Opsanus* spp.	132	0.16
Silk Snapper	*Lutjanus vivanus*	100	0.12
Night Shark	*Carcharhinus signatus*	74	0.09
Snakefish	*Trachinocephalus myops*	72	0.09
Crab, Unidentified	*Brachyura*	71	0.09
Coral	*Anthozoa*	64	0.08
Lionfish	*Pterois* spp.	60	0.07
Unidentified Bottom Debris	Unidentified Bottom Debris	60	0.07
Banded Rudderfish	*Seriola zonata*	56	0.07
Dolphin Fish (Mahi mahi)	*Coryphaena hippurus*	56	0.07
Queen Snapper	*Etelis oculatus*	56	0.07
Scalloped Hammerhead	*Sphyrna lewini*	54	0.07
Black Grouper	*Mycteroperca bonaci*	53	0.06
Sponge	*Porifera*	51	0.06
Blackfin Snapper	*Lutjanus buccanella*	48	0.06
Scorpionfish, Spiny Cheek	*Neomerinthe hemingwayi*	48	0.06
Carcharhinid, Unidentified	*Carcharhinus* spp.	42	0.05
Knobbed Porgy	*Calamus nadosus*	40	0.05
Sand Perch	*Diplectrum formosum*	37	0.04
Finfish, Unidentified	Unidentified Teleost	36	0.04
Cobia	*Rachycentron canadum*	35	0.04
King Snake Eel	*Ophichthus rex*	33	0.04
Yellowtail Snapper	*Ocyurus chrysurus*	32	0.04
Flatfish	*Pleuronectiformes*	26	0.03
Amberjack, Unidentified	*Seriola* spp.	24	0.03
Squirrelfish	*Holocentrus adscensionis*	24	0.03
Wahoo	*Acanthocybius solanderi*	24	0.03
Tilefish, Sand	*Malacanthus plumieri*	23	0.03
King Mackerel	*Scomberomorus cavalla*	22	0.03
Spinner Shark	*Carcharhinus brevipinna*	21	0.03
Starfish	*Asteroidea*	21	0.03
Sixgill Shark	*Hexanchus* spp.	18	0.02
Bank Sea Bass	*Centropristis ocyurus*	17	0.02

Table continued

Table 1.—*Continued.*

Common name	Scientific name	Number caught	Relative frequency (%)
Invertebrate, Unidentified	Invertebrate	17	0.02
Margate	*Haemulon album*	17	0.02
Blackbelly Rosefish	*Helicolenus dactylopterus*	15	0.02
Skates or Rays, Unidentified	*Batoidea*	15	0.02
Grouper, Unidentified	*Epinephelinae*	14	0.02
Great Hammerhead	*Sphyrna makarran*	11	0.01
Jack Crevalle	*Caranx hippos*	11	0.01
Lemon Shark	*Negaprion breirostris*	11	0.01
Unidentified	Unidentified	10	0.01
Warsaw Grouper	*Epinephelus nigritus*	10	0.01
Tilefish, Blackline	*Caulolatilus cyanops*	9	0.01
Bearded Brotula	*Brotula barbata*	7	0.01
Blacktip Shark	*Carcharhinus limbatus*	7	0.01
Conger Eel	*Conger oceanica*	7	0.01
Dusky Shark	*Carcharhinus obscurus*	7	0.01
Snapper, Unidentified	*Lutjanidae*	7	0.01
Swordfish	*Xiphias gladius*	7	0.01
Barrelfish	*Hyperoglyphe perciformis*	6	0.01
Common Octopus	*Octopus vulgaris*	6	0.01
Hammerhead, Unidentified	*Sphyrna* spp.	6	0.01
Goliath Grouper	*Epinephelus itajara*	5	0.01
Littlehead Porgy	*Calamus proridens*	5	0.01
Loggerhead Sea Turtle	*Caretta caretta*	5	0.01
Porgy, Unidentified	*Sparidae*	5	0.01
Rock Hind	*Epinephelus adscensionis*	5	0.01
Brown Pelican	*Pelecanus occidentalis*	4	<0.01
Cubera Snapper	*Lutjanus cyanopterus*	4	<0.01
Scorpionfish, Longspine	*Pontinus longispinis*	4	<0.01
Wenchman	*Pristipomoides aquilonaris*	4	<0.01
African Pompano	*Alectis ciliaris*	3	<0.01
Bull Shark	*Carcharhinus leucas*	3	<0.01
Chain Catshark	*Scyliorhinus retifer*	3	<0.01
Gull, Unidentified	*Laridae*	3	<0.01
Lobster, Unidentified	*Achelata*	3	<0.01
Scorpionfish, Spotted	*Scorpaena plumieri*	3	<0.01
Short Bigeye	*Pristigenys alta*	3	<0.01
Yellowfin Tuna	*Thunnus albacares*	3	<0.01
Sharpnose Sevengill Shark	*Heptranchias perlo*	2	<0.01
Billfish, Unidentified	*Istiophoriformes*	2	<0.01
Cornetfish, Red	*Fistularia petimba*	2	<0.01
Grunt, Unidentified	*Haemulidae*	2	<0.01
Herring Gull	*Larus argentatus*	2	<0.01
Sailfish, Atlantic	*Istiophorus albicans*	2	<0.01
Skipjack Tuna	*Katsuwonus pelamis*	2	<0.01
Whitebone Porgy	*Calamus leucosteus*	2	<0.01
Yellowmouth Grouper	*Mycteroperca interstitialis*	2	<0.01
Angel Shark	*Squatina dumeril*	1	<0.01
Atlantic Bonito	*Sarda sarda*	1	<0.01
Bigeye	*Priacanthus arenatus*	1	<0.01
Bluefish	*Pomatomus saltatrix*	1	<0.01
Clearnose Skate	*Raja eglanteria*	1	<0.01
Coney Grouper	*Cephalopholis fulva*	1	<0.01
Crustacean, Unidentified	Crustacean	1	<0.01
Dog Snapper	*Lutjanus jocu*	1	<0.01
Gannet	*Morus bassanus*	1	<0.01
Goosefish	*Lophius americanus*	1	<0.01
Graysby Grouper	*Epinephelus cruentatus*	1	<0.01
Laughing Gull	*Leucophaeus atricilla*	1	<0.01
Longtail Bass	*Hemanthias leptus*	1	<0.01
Seabird, Unidentified	*Aves*	1	<0.01
Sheepshead	*Archosargus probatocephalus*	1	<0.01
Triggerfish, Unidentified	*Balistidae*	1	<0.01

In total, unidentified sharks accounted for a low percentage (6%).of the total shark catch.

Nearly 94% of the sharks arrived at the vessels alive, but some shark species were more adept at surviving on commercial BLL fishing gear than others, as reflected in Table 4. Notable shark species with high survival included blacknose (97%), sandbar (100%), and tiger, *Galeocerdo cuvier* (100%). In comparison, silky sharks, *Carcharhinus falciformis*, arrived at vessels dead 34% of the time, while the scalloped hammerhead, *Sphyrna lewini*, had an at-vessel mortality rate of 39%. Dusky sharks were the most likely to succumb from capture with a 43% mortality rate. Of the sharks that arrived at a vessel, only 8% were removed from the population by being discarded dead or retained as bait. Small coastal sharks, including the Atlantic sharpnose and blacknose, represented the majority of those retained as bait, as well as juvenile silky sharks. A total of 1% of all sharks were recorded as having an "unknown condition" at capture, and 0.5% that had an "unknown fate."

Among the shark bycatch, 56% had biological information documented through EM review, providing valuable data for further analysis. This included 30% whose sex was determined by the presence or absence of claspers, resulting in 755 females and 423 males. Based on size range, there were 405 juveniles and 645 identified as an adult. Sharks classified as small (<1 m) included juveniles, deepwater sharks, and small coastal species, comprising 42% of the shark catch. Large sharks (>2 m) were mainly adults and accounted for 18% of all recorded sharks.

Examining all shark species collectively, hotspots of CPUE were identified in the extreme northwest and southeastern corners of the fishing area (Fig. 14). However, extensive clustering patterns were not observed for most of the entire fishing area, resulting in no cold spots being observed and indicating a predominantly random distribution. Two complete and one partial hotspot occurred at greater depths in the northwestern area centered near lat. 28°94'N and long. 86°00'W, driven by dogfish shark species catches, with one additional cell occurring due east in shallower depths. The southeastern cluster of two grid cells was centered near lat. 24°78'N, long. 83°12'W encompassing 755 km^2, and represented a variety of different species encountered. When all shark species were grouped, CPUE remained relatively stable, ranging from approximately 1.0 to 1.3 indi-

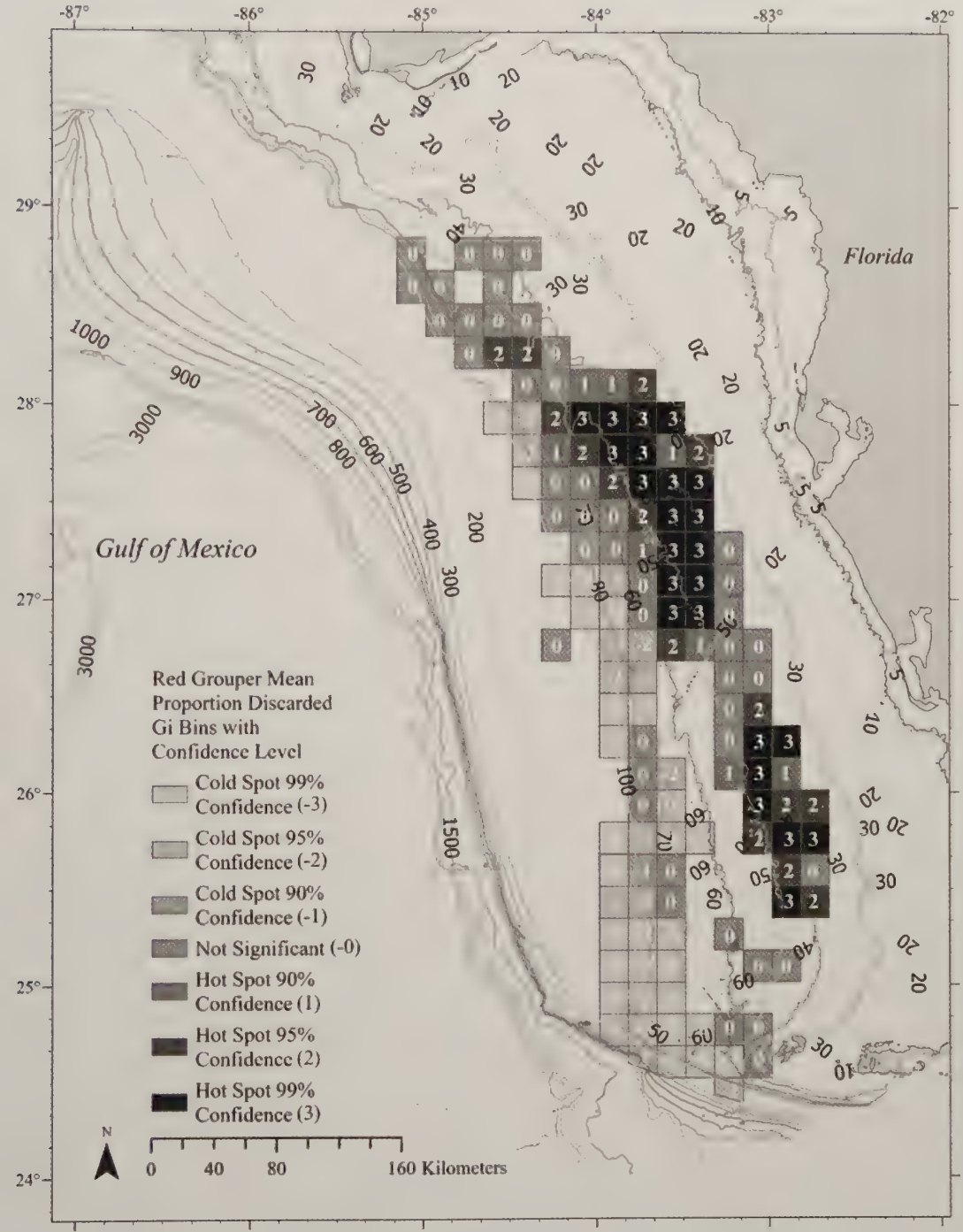

Figure 4.—Spatial clusters of mean high value (hotspot), low value (coldspot), and random distributions of the proportion of landed red grouper discarded at sea spatially joined to 10.0-min grids from bottom longline vessels fishing the eastern Gulf of Mexico from July 2016 through December 2021. Isobaths are shown in meters.

Table 2.—Condition on arrival of red grouper, red snapper, and other bony fishes caught by participating bottom longline vessels.

Condition	No. of red grouper	% of red grouper	No. of red snapper	% of red snapper	No. of other bony fishes	% of other bony fishes
Dead - damaged	406	0.76	158	2.21	317	1.79
Dead - undamaged	516	0.96	403	5.63	655	3.69
Live - damaged	134	0.25	13	0.18	49	0.28
Live - healthy	29,887	55.61	5,099	71.27	9,945	56.04
Live - stomach and/or eyes protruding	22,635	42.12	1,473	20.59	6,643	37.43
Unknown condition	162	0.30	8	0.11	137	0.77

Table 3.—Fate (disposition) of red grouper, red snapper, and other bony fishes caught by participating bottom longline vessels.

	Red grouper		Red snapper		Other bony fishes	
Catch and fate	Number	%	Number	%	Number	%
Discarded – dead	956	1.78	461	6.44	708	3.99
Discarded – live and damaged (not vented)	97	0.18	16	0.22	25	0.14
Discarded – live and damaged (vented)	132	0.25	6	0.08	3	0.02
Discarded – live and healthy (not vented)	7,510	13.97	592	8.28	1,045	5.89
Discarded – live and healthy (vented)	17,004	31.64	579	8.09	232	1.31
Discarded – unknown	120	0.22	6	0.08	77	0.43
Retained	27,814	51.76	5,486	76.68	13,526	76.22
Retained as bait	3	0.01	1	0.01	1,940	10.93
Unknown fate	104	0.19	7	0.10	190	1.07

Table 4.—Condition of shark catch by species based on percentage arriving to the vessels dead as identified through electronic monitoring.

Common name	Scientific name	Number caught	Number dead	% Dead on arrival
Dusky Shark	*Carcharhinus obscurus*	7	3	42.86
Scalloped Hammerhead	*Sphyrna lewini*	54	21	38.89
Silky Shark	*Carcharhinus falciformis*	148	51	34.46
Great Hammerhead	*Sphyrna makarran*	11	3	27.27
Spinner Shark	*Carcharhinus brevipinna*	21	3	14.29
Night Shark	*Carcharhinus signatus*	74	9	12.16
Carcharhinid, Unidentified	*Carcharhinus* spp.	42	4	9.52
Atlantic Sharpnose Shark	*Rhizoprionodon terraenovae*	912	64	7.02
Shark, Unidentified	*Selachimorpha* spp.	183	11	6.01
Blacknose Shark	*Carcharhinus acronotus*	514	16	3.11
Dogfish, Smooth	*Mustelus* spp.	428	4	0.93
Dogfish, Spiny	*Squalidae* spp.	378	2	0.53
Sandbar Shark	*Carcharhinus plumbeus*	597	2	0.34
Sharpnose Sevengill Shark	*Heptranchias perlo*	2	0	0.00
Angel Shark	*Squatina dumeril*	1	0	0.00
Blacktip Shark	*Carcharhinus limbatus*	7	0	0.00
Bull Shark	*Carcharhinus leucas*	3	0	0.00
Chain Catshark	*Scyliorhinus retifer*	3	0	0.00
Hammerhead, Unidentified	*Sphyrna* spp.	6	0	0.00
Lemon Shark	*Negaprion breirostris*	11	0	0.00
Nurse Shark	*Ginglymostoma cirratum*	237	0	0.00
Sixgill Shark	*Hexanchus* spp.	18	0	0.00
Tiger Shark	*Galeocerdo cuvier*	316	0	0.00
Total		3,973	193	

viduals per 1,000 hook hours (Fig. 15); small-bodied shark species contributed disproportionately to CPUE during the earlier half of the study.

Sandbar sharks were the most predominant large coastal shark and were caught at depths between 37 m and 243 m, and an average depth of 66 m. High confidence CPUE hotspots were aggregated into two groups, one group of three highly significant grid cells occurred in the northwest end of the fishing area centered near lat. 28°43'N, long 84°68'W, and three cells totaling 1,132 km^2, with a proximal cell located further northwest. The second group was at the extreme southwestern end of the fishing area, proximal to the Dry Tortugas and Pulley Ridge Marine Protected Areas, and was centered around lat. 24°68'N, long. 83°45'W with four complete and three partial cells (Fig. 16). Though no seasonal trends were apparent, sandbar sharks were the only shark species that showed an increase in CPUE, approximately 66% from 2016 to 2021 (Fig. 15). In comparing sandbar shark hotspots CPUE (Fig. 16) and those of recorded depredated fish species (Fig. 17), the hotspots did not align to validate if this was the primary shark species driving depredation.

Sea Turtle and Seabird Interactions

Sea turtles and seabirds represented a small portion of the incidental bycatch. Among the five loggerhead turtles, *Caretta caretta,* encountered, two mortalities were recorded. Interestingly, there were no recorded encounters in the years 2019 and 2020. Seasonally, two sea turtles were captured in winter months (January–March), two in the spring (April–June), and one in the fall (October–December), with no captures recorded during the summer months (July–September). The incidental cap-

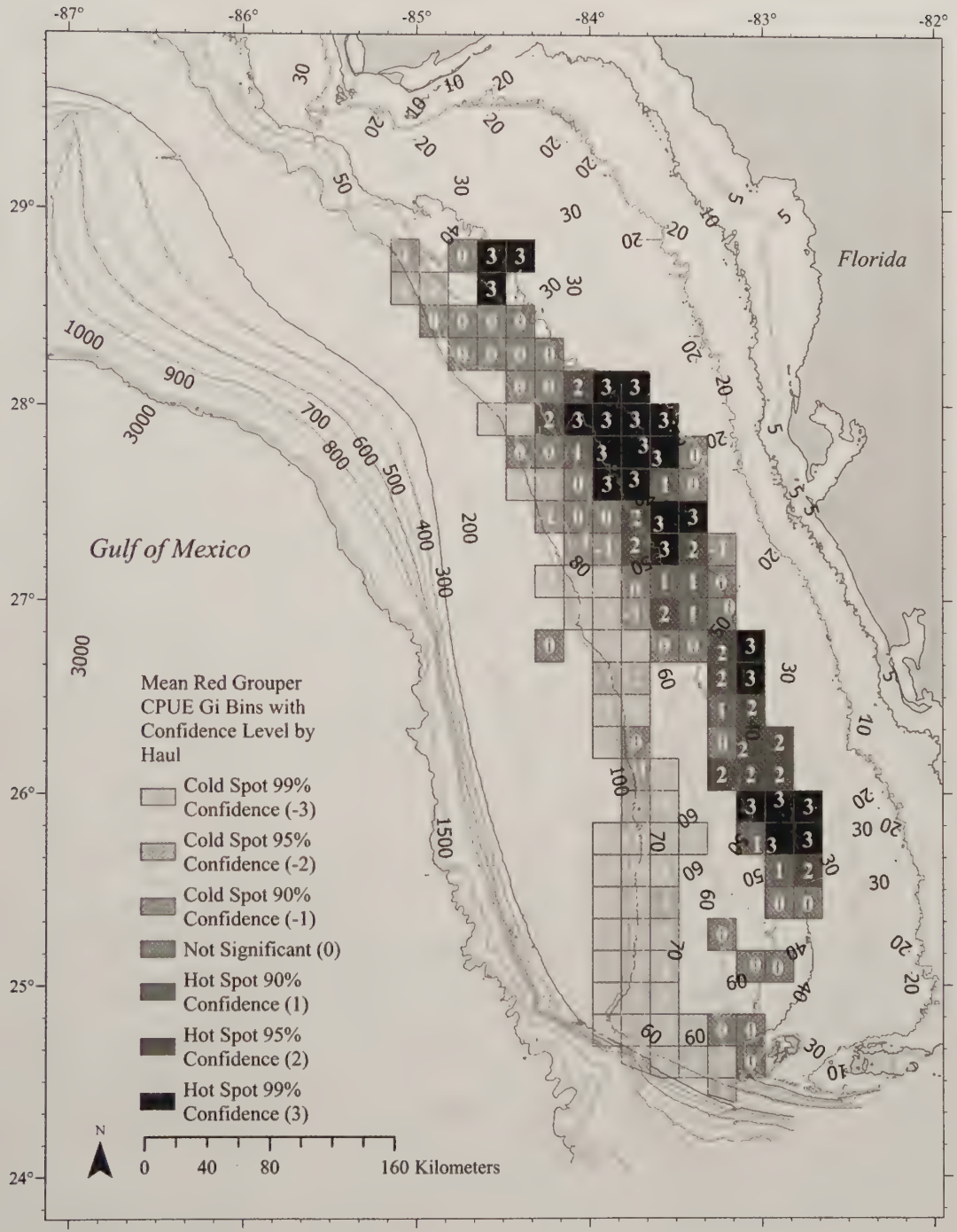

Figure 5.—Spatial clusters of red grouper catch depicted as mean high value (hotspot), low value (coldspot), and random distributions of species-specific catch per unit effort (set-haul hook-hours x 1,000) spatially joined to 10.0-min grids. Isobaths are shown in meters.

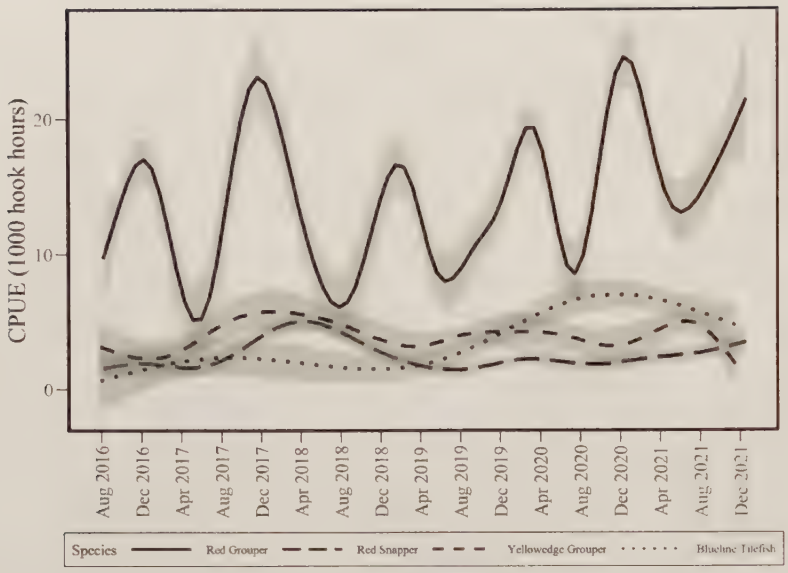

Figure 6.—Catch per unit effort (CPUE) trends for red grouper, red snapper, yellowedge grouper, and blueline tilefish catch.

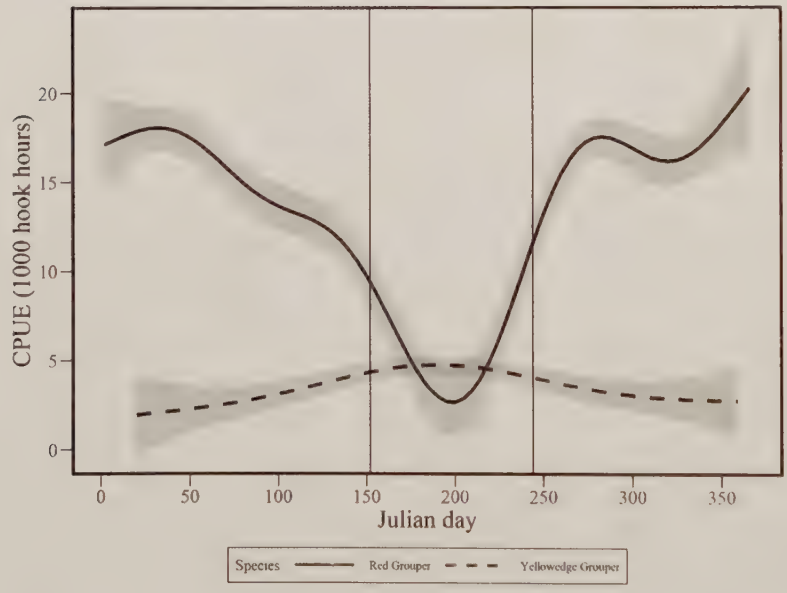

Figure 7.—Catch per unit effort (CPUE) for red grouper and yellowedge grouper catch by day of year relative to the June through September closure inside 35 fathoms (vertical lines).

ture of birds resulted in a high mortality rate of 90%, with various gulls, *Laridae* being the most affected species (n = 6), followed by brown pelicans, *Pelecanus occidentalis* (n = 4), and one gannet, *Morus bassanus*.

Discussion

The application of EM over a span of five years in partnership with the eastern U.S. Gulf BLL reef fish fishery, utilizing Saltwater Inc, EM hardware and software, yielded promising results. This initiative facilitated the collection of detailed fishing activity observations and comprehensive documentation of catch, bycatch, and particularly discard species, along with their quantity, location, and subsequent disposition. Through the voluntary efforts of the fishing industry, a significant milestone of over 20% of the eastern Gulf BLL fleet was participating in EM coverage as of 2021. During the study period from the summer of 2016 through 2020, EM systems documented 306 BLL trips, nearly three times the number of trips covered by the regulatory reef fish observer program (105) for all Gulf BLL vessels (Atkinson[14]). This additional stream of data is particularly important for management consideration, given this area's high species diversity (Pulver and Stephen, 2019), limited (~2%) at-sea observer coverage (Scott-Denton[4]), and challenges associated with the unreliable self-reported discard data by fishermen (GMFMC[5]). Gaining a better understanding and accurately documenting discarded unintentional or non-targeted catch is especially critical in fisheries management. These discards, often comprising a significant proportion of the catch (Harrington et al., 2005), are characterized by high release mortality rates, influenced by various factors (Shertzer et al., 2021). Therefore,

[14]Atkinson, S, S. G. Smith, and G. Decossas. 2023. CPUE Expansion estimation for commercial discards of Gulf of Mexico yellowedge grouper (*Hyporthodus flavolimbatus*). SEDAR85-WP-06. SEDAR, North Charleston, SC, 28 p. (https://sedarweb.org/documents/sedar-85-wp-06-cpue-expansion-estimation-for-commercial-discards-of-gulf-of-mexico-yellowedge-grouper-hyporthodus-flavolimbatus/).

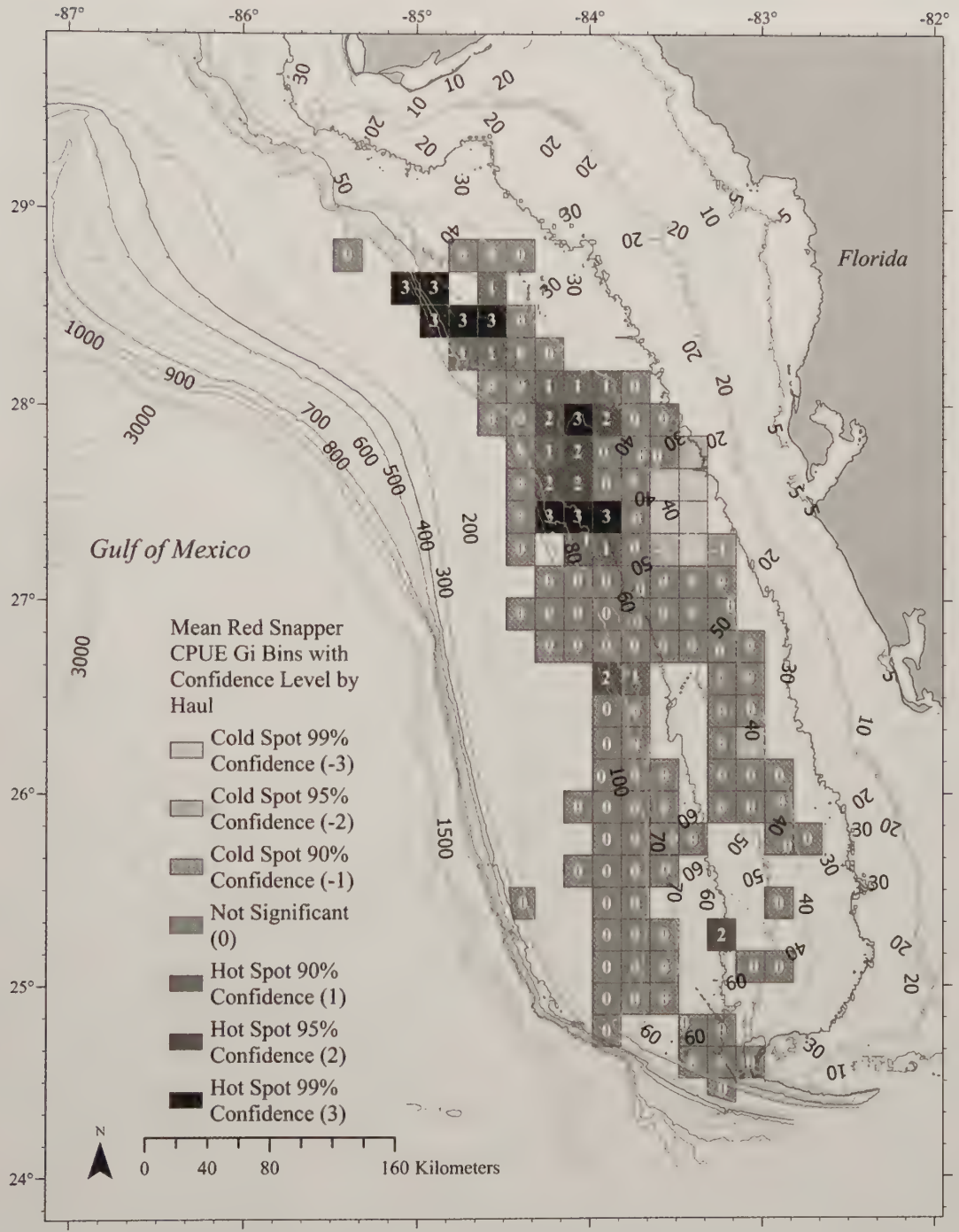

Figure 8.—Spatial clusters of red snapper catch depicted as mean high value (hotspot), low value (cold spot), and random distributions of species-specific catch per unit effort (set-haul hook-hours x 1,000) spatially joined to 10.0-min grids. Isobaths are shown in meters.

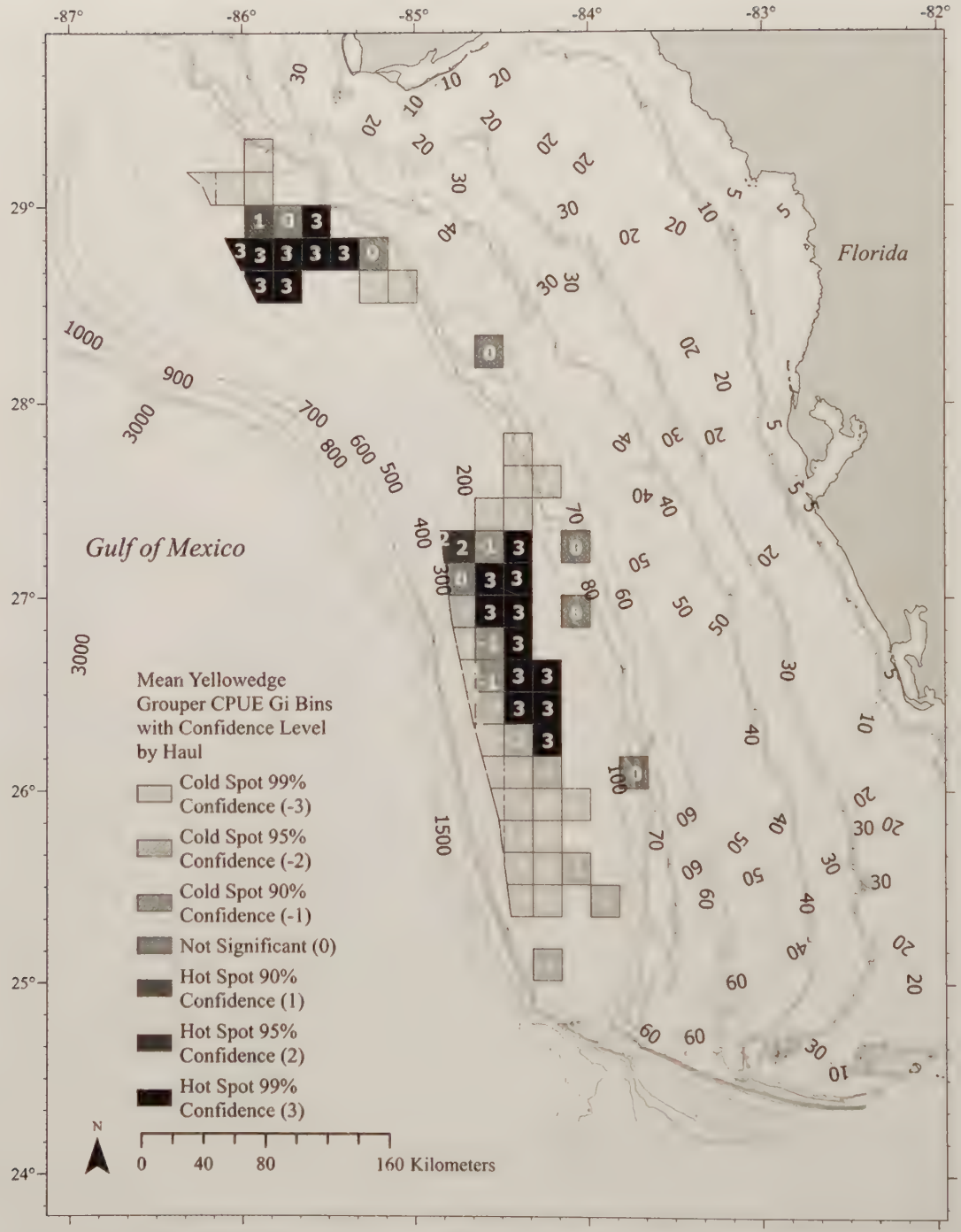

Figure 9.—Spatial clusters of yellowedge grouper catch depicted as mean high value (hotspot), low value (coldspot), and random distributions of species-specific catch per unit effort (set-haul hook-hours x 1000) spatially joined to 10.0-min grids. Isobaths are shown in meters.

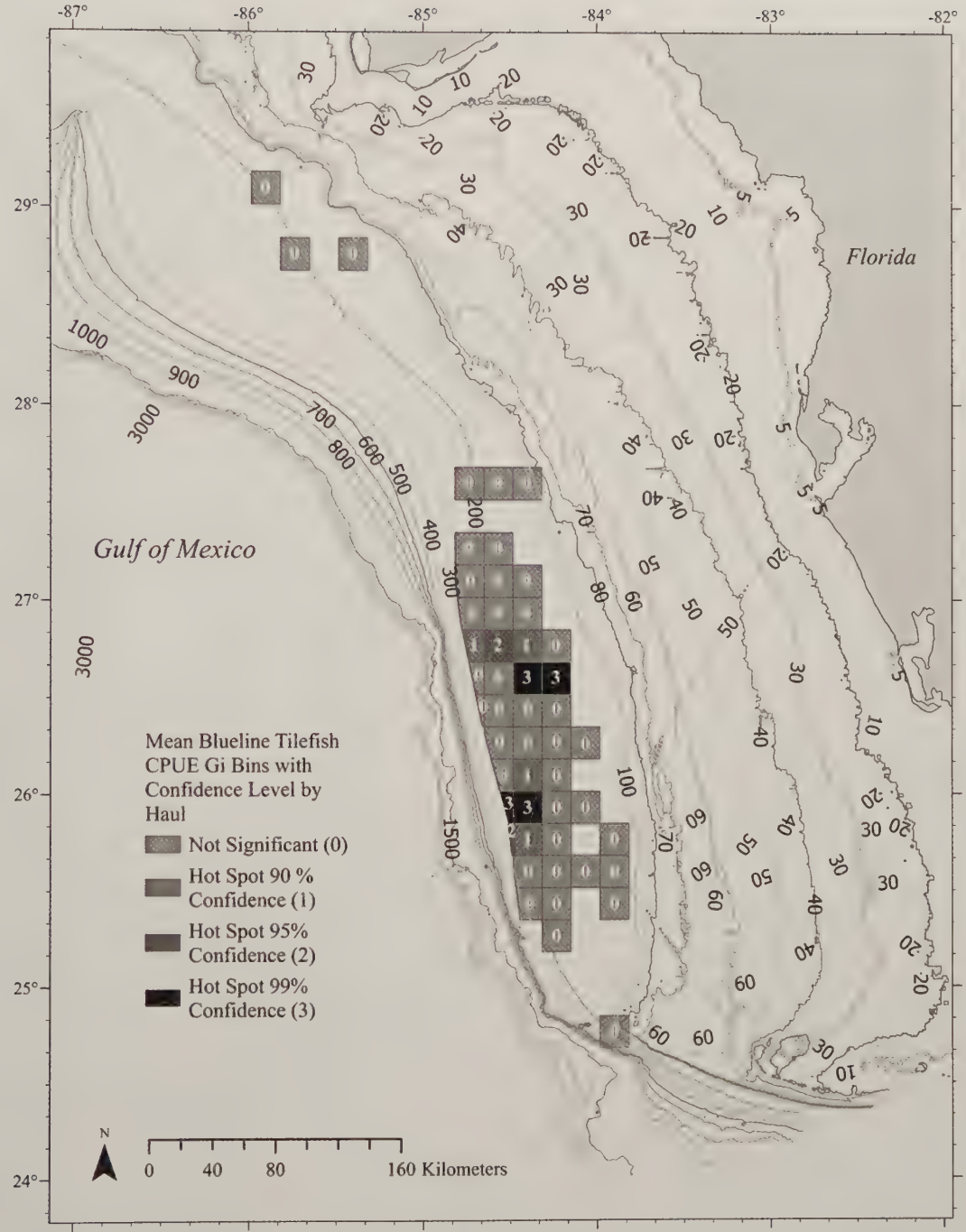

Figure 10.—Spatial clusters of blueline tilefish catch are depicted as mean high value (hotspot), low value (cold spot), and random distributions of species-specific catch per unit effort (set-haul hook-hours x 1,000) spatially joined to 10.0-min grids. Isobaths are shown in meters.

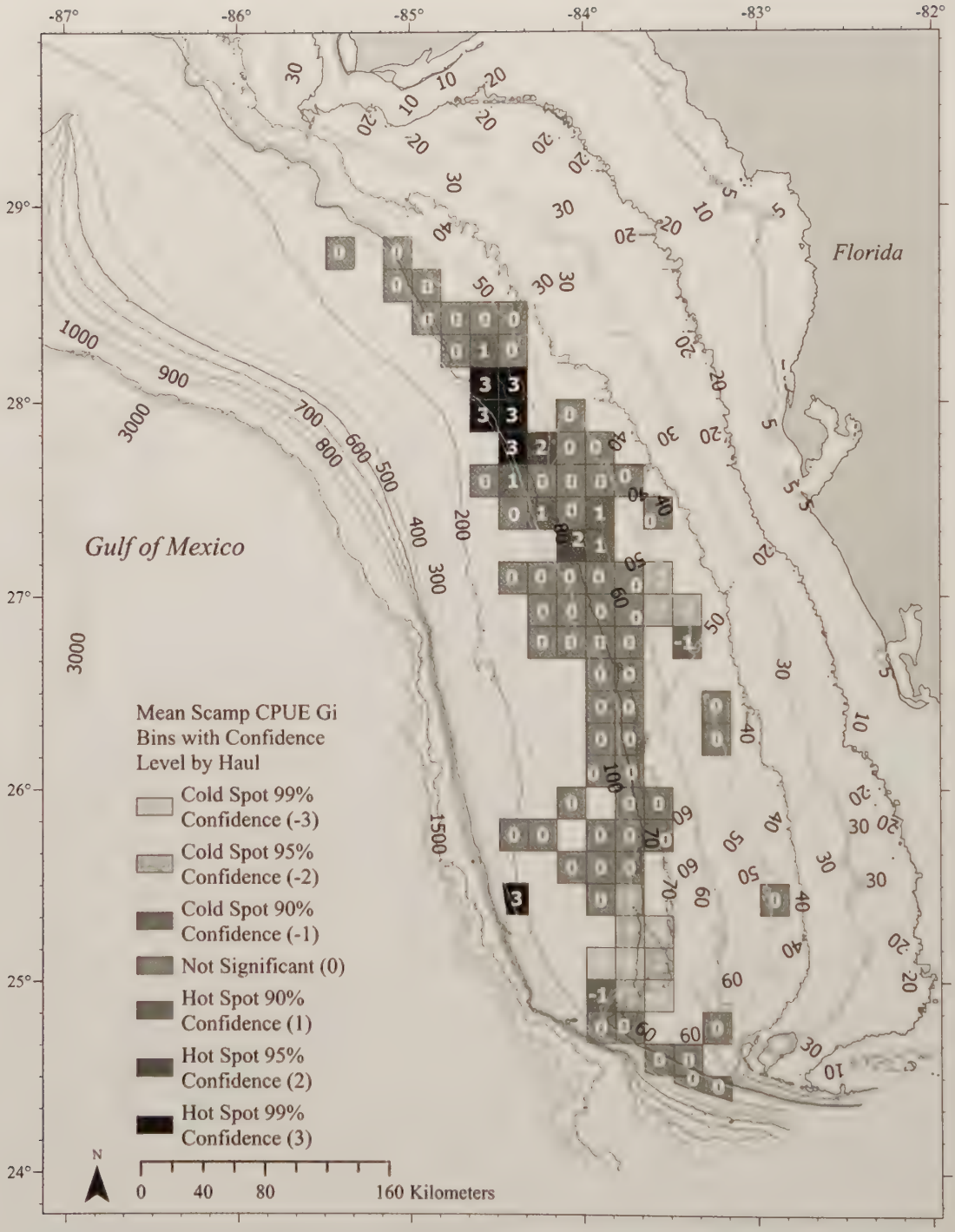

Figure 11.—Spatial clusters of scamp catch depicted as mean high value (hotspot), low value (cold spot), and random distributions of species-specific catch per unit effort (set-haul hook-hours x 1000) spatially joined to 10.0-min grids. Isobaths are shown in meters.

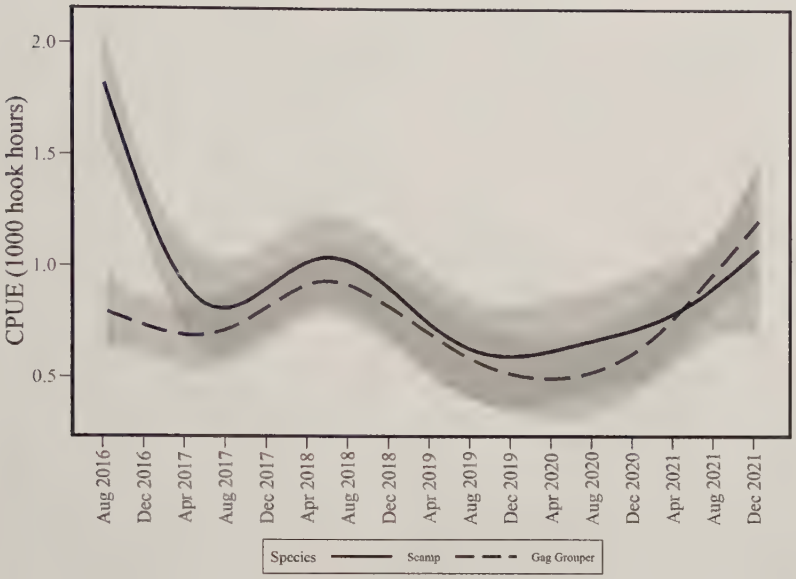

Figure 12.—Catch per unit effort (CPUE) as related to 1,000 hook hours for scamp and gag grouper catch from bottom longline vessels.

implementing solutions to enhance and augment the accuracy of discard documentation is essential for generating reliable stock assessments (Suuronen and Gilman, 2020) and informing management practices.

Our findings align with previous research that has assessed the efficacy of video monitoring, and affirms that EM can serve as a reliable and accurate method to estimate catch (McElderry et al., 2003; Ames et al., 2007; Stanley et al., 2011; van Helmond et al., 2014, 2020). This is especially notable in fisheries challenged by low observer coverage, as acknowledged in studies by Michelin et al. (2018), Bradley et al. (2019), and Gilman et al. (2019). Following these investigations, our study emphasized improving views through camera positioning, including use of boom mounts, to view catches in a way that allowed individual fish to be tracked from their arrival to their disposition. This level of monitoring was also achieved through ongoing communications with the vessel captains and crew to ensure technical issues with the EM systems and any changes in fish processing locations were addressed quickly.

The use of EM technology is rapidly gathering momentum, and can perform some functions at a higher level than human observers, though there can be challenges such as poor lighting or dirty camera domes, fish handled out of frame, high volume of fish across the camera frame (Ruiz et al., 2015; van Helmond et al., 2020; Sylvia[15]; NMFS[16]), fish released while underwater, and/or EM system malfunction. This technology has demonstrated suitability for use across a wide range of vessels, providing the capability to review video for data verification, increase scalability, and engage industry in the self-reporting processes (van Helmond et al., 2020). Relative to human observers, EM programs are typically lower in cost, and this advantage is expected to expand with ongoing advancements in technology (Michelin et al., 2018). Although EM programs traditionally do not evaluate the size structure and gear specificity like traditional observer programs, integrating EM with clearly established standards (Stanley et al., 2011, 2015) into fishery-dependent monitoring programs can contribute to assessments of exploited stocks (ICES, 2019). It can also aid in the quantification of impacts on species, particularly endangered, threatened, and protected species (Gray and Kennelly, 2018). Furthermore, EM integrated as new data stream can contribute valuable insights into vessel and fleet operations, fishing effort, and information concerning discards and discarding practices (Bradley et al., 2019; van Helmond et al., 2020; Suuronen and Gilman, 2020).

Gulf reef fish fishery vessel owners and captains voluntarily participated in this work due to their expressed interest to have the permanent video documentation and the resulting data analysis. They aimed to leverage this data to improve their fishing operations, demonstrate to fisheries managers that they were applying sustainable fishing practices, and to contribute additional accurate information for managers to consider in making more informed decisions regarding regulations for this fishery. Similar motivations for enhancing stock assessments and promoting sustainability were also reported in studies by van Helmond (2020) and Plet-Hansen et al. (2017). Many participants expressed a preference for applying EM full-time over occasional at sea-observer coverage. During the period of 2016 through 2021, observers were present on roughly 3% of the EM trips. Interestingly, some captains expressed a willingness to adopt EM systems if NMFS granted an observer waiver. This suggests a potential interest for applying EM technology as an alternative for traditional at-sea observer coverage. However, effectively implementing EM as an alternative for these exemptions would necessitate coordination with both management and the vessel captains to ensure compliance.

[15]Sylvia, G., M. Harte, and C. Cusack. 2016. Challenges, opportunities, and costs of electronic fisheries monitoring. The Environmental Defense Fund, San Francisco, CA,. 34 p. (https://www.edf.org/sites/default/files/electronic_monitoring_for_fisheries_report_-_september_2016.pdf).

[16]NMFS Office of Policy and Electronic Monitoring Working Group. 2013. Electronic monitoring and electronic reporting: guidance and best practices for federally-managed fisheries (http://www.nmfs.noaa.gov/op/snippets/em_er_discussion_draft_august_2013.pdf)

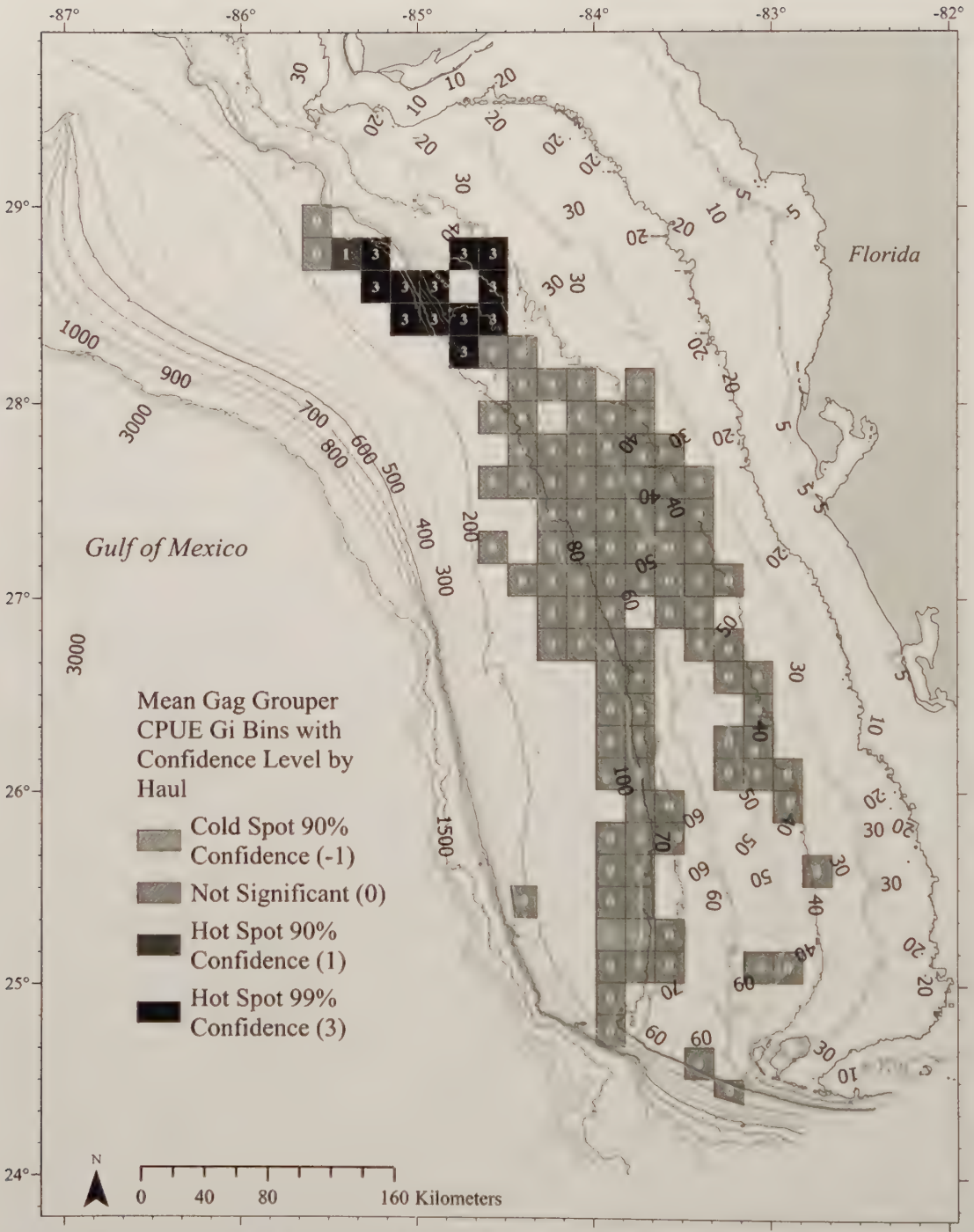

Figure 13.—Spatial clusters of gag grouper catch depicted as mean high value (hotspot), low value (coldspot), and random distributions of species-specific catch per unit effort (set-haul hook-hours x 1000) spatially joined to 10.0-min grids. Isobaths are shown in meters.

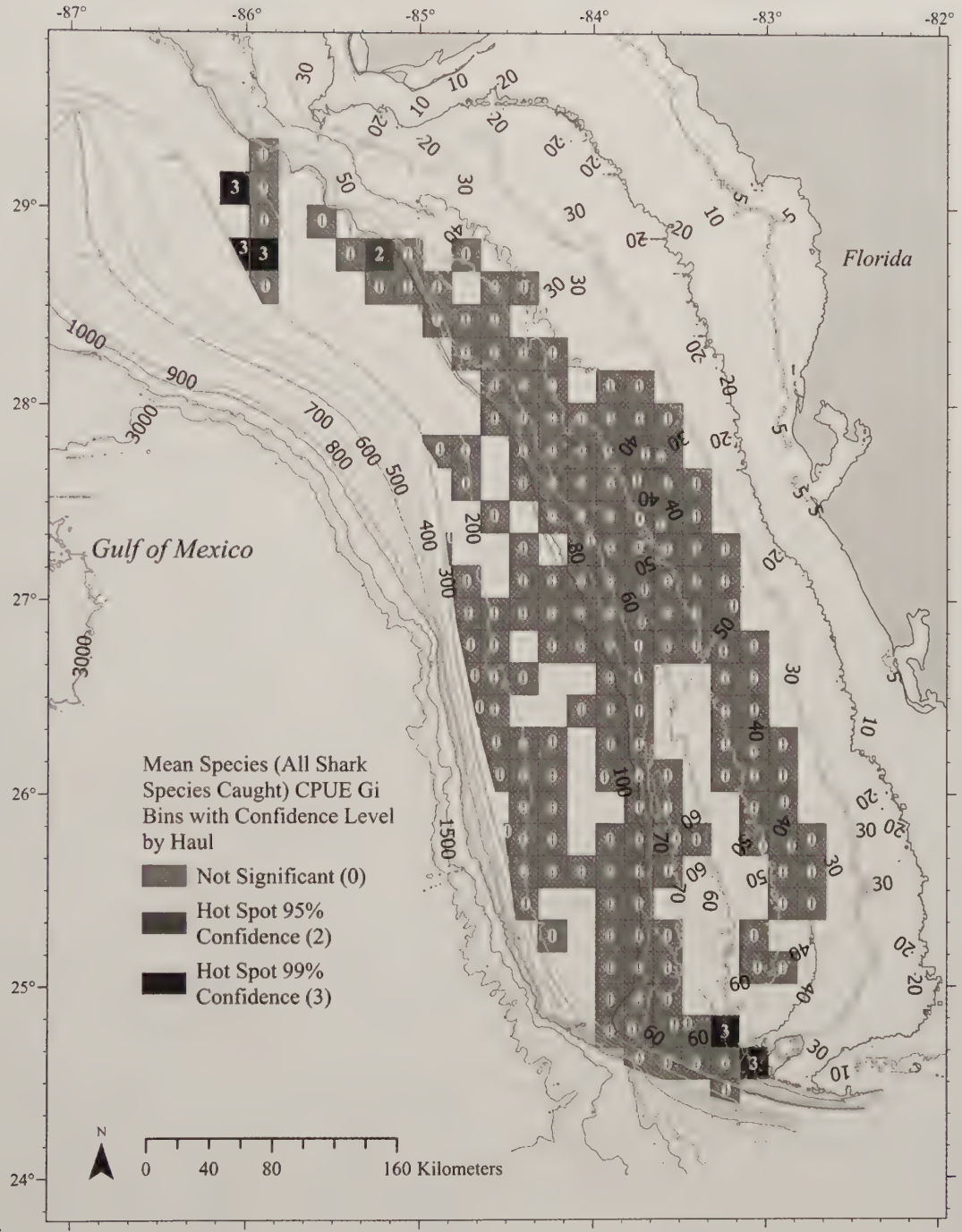

Figure 14.—Spatial clusters of catch of all shark species depicted as mean high value (hotspot), low value (coldspot), and random distributions of species-specific catch per unit effort (set-haul hook-hours x 1,000) spatially joined to 10.0-min grids. Isobaths are shown in meters.

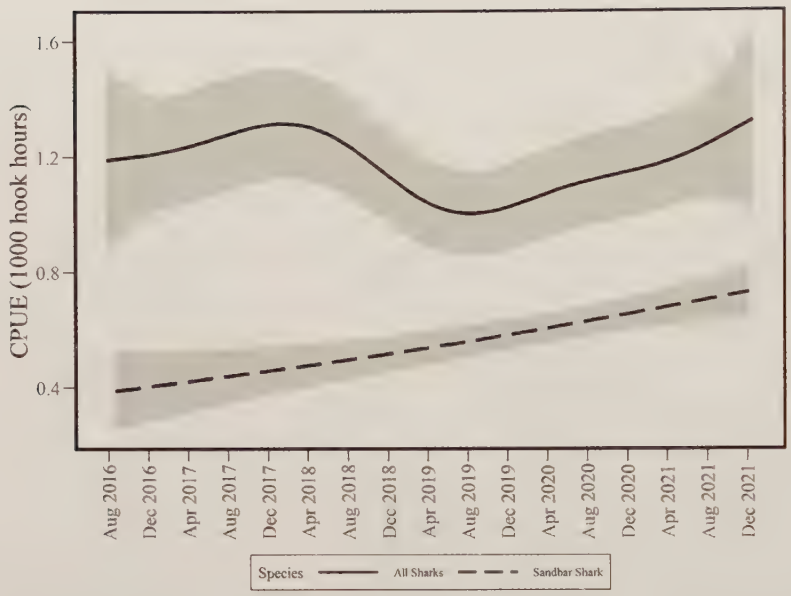

Figure 15.—Catch per unit effort (CPUE) as related to 1,000 hook hours for all sharks and sandbar shark catches from bottom longline vessels.

Several vessel owners and captains who expressed their willingness to participate in EM data collection, cited their trust in the work being carried out by a non-governmental and non-enforcement organization. However, there were also fishermen who declined to take part, citing concerns over potential privacy intrusion. In addition, several captains expressed a general distrust in any form of monitoring, and voiced concerns that broader data collection efforts might result in further restrictions being imposed. Efforts were made to address these concerns to the best of our ability, emphasizing the potential benefits of more informed management for the fishery, which would ultimately serve the interests of the fishermen. Similar sentiments, highlighting concerns among some fishermen regarding potential changes in data collection and management practices in fisheries, were reported by Mangi et al. (2013), Eayrs et al. (2015), and Plet-Hansen et al. (2017) in their on-water camera surveillance research studies and in fishery compliance applications.

There were some challenges in maintaining consistency with vessels due to unexpected extended periods in port for major repairs, sales, or relocations to distant ports. These challenges required persistent and constant communications with industry representatives to make every effort to maintain consistent and reliable data collection. While there were several captains who worked the same vessel for the full duration, turnover among captains and particularly crew members was common across vessels. In several instances, vessels experienced turnover with more than five different captains during the project period. These situations provided the opportunity to gather data on the variability in effort and catch on individual vessels across multiple captains over time, a task that a limited number of human observers may not have been able to accomplish.

Many vessels, which had been actively fishing for decades, exhibited ongoing power-related issues stemming from inadequate wiring or the use of aged marine batteries. While captains were assured of the low amperage draw of the EM system, some expressed concerns that they might exhaust their battery power after dusk, as their battery draw increased with multiple deck lights illuminating fishing activities. In some cases, an integrated battery backup or replacement marine battery was provided. Few vessels used generators, which supplied a more reliable power source. Reliance on volunteer captains and aged vessels did not detract from the program's ability to gather fairly consistent catch, bycatch, and discard data for producing data products in support of industry and management goals.

The Saltwater Inc. EM vessel monitoring systems, including the two camera models, were found to be durable and long-lasting under the adverse conditions presented by the Gulf climate. The only irreparable damage occurred to one processor due to saltwater intrusion during a severe storm. Replacement of components, such as monitors, system power packs, and drum sensor hardware, was typically due to wear and tear over time. The preferred camera model was the GeoVision Model GV-EVD3100, due to simply removing the outer face plate to make minor vertical or horizontal adjustments to the lens to improve views. In contrast, making minor lens positioning adjustments on the Vivotek model was challenging, as it required unmounting the camera from the vessel to access the internal components through the base plate and then remounting to confirm the lens adjustment was correct. Additionally, a point of concern with the Vivotek camera model was the use of an external rather than internal cable coupler. This was a concern for possible moisture intrusion or the camera cable becoming disconnected. As a preventive measure, cable connections were bound with marine grade tape and secured to make a drip loop. Regardless of the camera brand, occasional poor image quality was observed due to dried salt spray, clinging heavy dew or rain drops, or fish slime. Minor scratches sometimes occurred on the domes, but they were typically not problematic unless they were directly in front of the lens. Captains and their crew were encouraged to clean

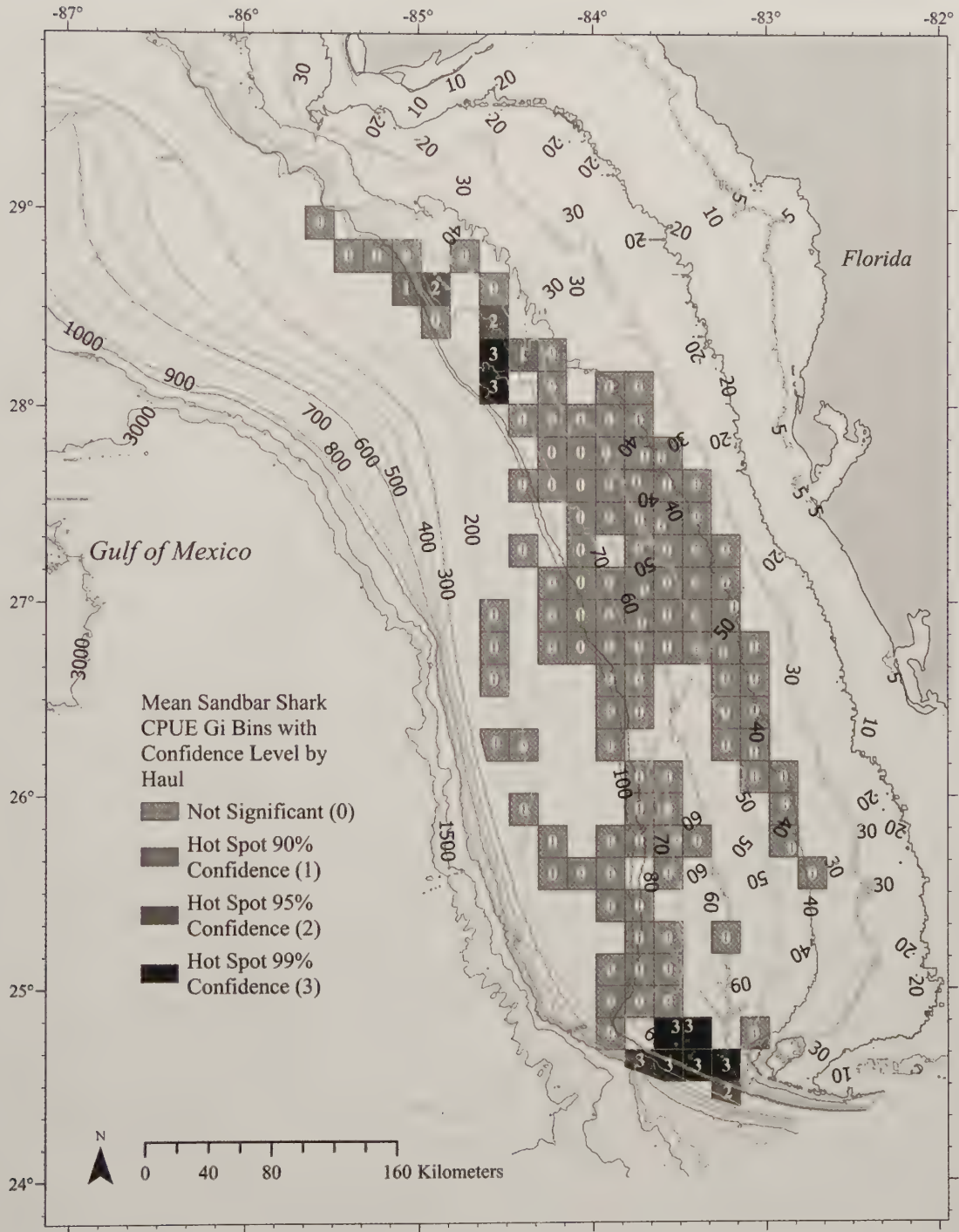

Figure 16.—Spatial clusters of sandbar sharks mean depicted as high value (hotspot), low value (coldspot), and random distributions of species-specific catch per unit effort (set-haul hook-hours x 1,000) spatially joined to 10.0-min grids. Isobaths are shown in meters.

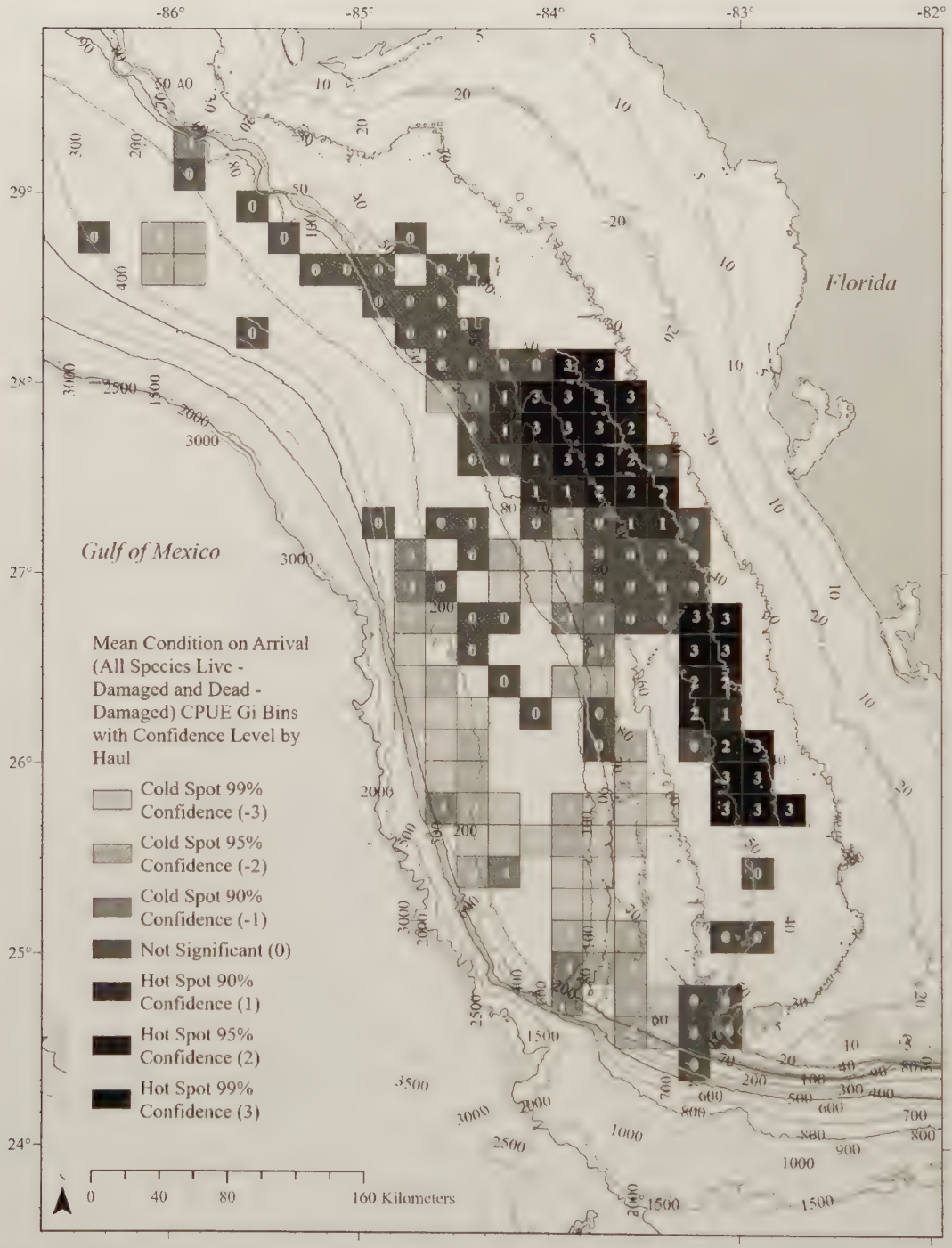

Figure 17. Spatial clusters of depredated fish mean depicted as high value (hotspot), low value (coldspot), and random distributions of species-specific catch per unit effort (set-haul hook-hours x 1,000) spatially joined to 10.0-min grids. Isobaths are shown in meters.

dirty camera domes and were provided with cleaning supplies.

Cameras mounted on vessel specific aluminum booms were integrated early during this work and made a significant improvement in views of the longline gear and fish as they were hauled and brought onboard or were released in close proximity to the vessel. Booms also improved views for the identification of large sharks that were discarded at the rail while underwater near the vessels and for documenting at-surface predation of discards by sharks or marine mammals.

Camera placement to maintain unobstructed views of the deck processing areas was critical for not only optimizing the documentation of individual fish from haul to discard, but also for reviewers to make detailed observations on fish condition and handling procedures, notably the presence of barotrauma and venting. It was noted that most crew members were aware of the benefits of venting and more often than not took the extra effort to do so; however, the vast majority of observed venting efforts were not performed correctly. This may have been attributed to crew turnover and a lack of hands-on training. During early monitoring, only a few fishermen used NMFS recommended hollow sharp cannula venting tools, with reviewers observing that filet knives and kill picks were most often the tools of choice. Unfortunately, these tools were often applied incorrectly, potentially causing internal damage. As reported by Drumhiller et al. (2014), improper venting techniques can decrease the survival odds for fish released with barotrauma.

In fisheries electronic monitoring, there is evidence to suggest that fishermen may adjust their discard behavior in response to the presence of monitoring systems, leading to changes in fishing practices, sorting techniques, or even avoidance of monitored areas or species. However, the extent of these adjustments can depend on factors such as the level of enforcement, the effectiveness of monitoring systems, and if economic incentives are involved. Interestingly, in this study, discard behavior did not seem markedly influenced across participating volunteer vessels. As we reported in this study, there were observed instances where bycatch, which should have been discarded, were repurposed as bait, in addition to instances of improper venting.

The most recent comparable fishery-dependent data for the Gulf BLL reef fish fishery is from the region's fisheries observer program, where 90% of the fishing effort recorded from 2006 to 2009 occurred specifically in the eastern Gulf (Scott-Denton et al., 2011). The observer and EM BLL vessel data results were reasonably consistent in identifying areas with high fishing effort. Based on EM data, the majority of effort, 61%, was recorded in zones 4 and 5 (Fig. 1), while information based on observer data reported zone 4 as having the highest fishing effort (35%) among zones 3 through 5. Red grouper was the predominant catch in the eastern Gulf. According to EM data, fishing effort in statistical zones 4 and 5 produced more than 80% of the total catch, which consisted of red grouper (65%) and red snapper (9%), followed by yellowedge grouper (5%), and blueline tilefish (4%) (Table 1). However, observer data from Scott-Denton et al. (2011) showed a lower contribution (56%) of red grouper, with red snapper catches accounting for only 3% of their observed harvest. It was reported that yellowedge grouper used to be the second most common species and comprised 10% of the catch, followed by blueline tilefish (5%), other tilefish, and Atlantic sharpnose sharks, which were each 3%.

Generally, the EM systems were able to generate some of the same types of data as an onboard observer, including fishing activity and catch and discard composition and disposition, and at times surpassed the capability of a human observer. For instance, in the case of shark identification, EM data showed 6% of sharks were recorded as "unknown" or "unknown carcharhinid," while early observer reports (Scott-Denton et al., 2011) for unknown shark groupings were higher (16%). This could potentially be due to various factors, including observers biological sampling workload, inexperience, not being able to observe crew releasing sharks while underwater, and possibly obstructed views.

Applying EM in this fishery not only was useful for documenting retained catch, but more importantly for documenting discarded species and their disposition (dead or live). Usually, fishing pressure has been monitored using only estimates of the portion of the catch that fishermen retain (Gilman et al., 2019). Because discards can be substantial, optimal camera coverage in this study was beneficial for monitoring and accounting for individual fish and their disposition. As emphasized by Cook (2019), Gilman et al. (2019), and Suuronen and Gilman (2020), comprehensive data is essential for estimating total fishing mortality to support more robust stock assessments and improved fisheries management.

High discard rates observed in the Gulf commercial reef fish fishery are attributed to regulations based on size limits, quota restrictions, and marketability (Pulver and Stephen, 2019). According to EM collected data, nearly half of all of the red grouper caught were discarded, primarily due to size limit restrictions. Minimal yellowedge discards were attributed to fishermen who had sufficient quota when targeting deepwater species, which promoted management related discarding. Conversely, according to the captains, red snapper discards were primarily attributed to quota related issues or fishermen's concern over the quality and marketability of those fish that arrived at the vessel dead with discolored gills, rather than size limits. Marketability was also a driver of blueline tilefish discards, as they have no size limit, but low prices leave fishermen with limited profits after paying for leased quota compared to other reef fish species (Pulver and Stephen, 2019). Management or market-based solutions are needed to improve blueline tilefish sustainability, as discard mortality studies show an estimate of post-release

mortality of 90%, with the potential for discard mortality to be 100% (SEDAR[17]). Limited discards observed for the other economically important reef fish species, including scamp and gag grouper, were attributed to no size limit restriction, or that they were generally a larger size class of fish caught by this gear type. As reported by Suuronen and Gilman (2020), at-sea human observer programs produce accurate discard data. However, the potential of EM as a highly effective method is highlighted due to its ability to overcome most sources of statistical sampling bias in conventional human onboard observer programs. Our study showed that EM is capable of providing extensive discard data. Moving forward, it is important to take steps towards constructing a management framework that integrates EM into the larger data collection process. Through the adoption of an integrated approach, the full potential of EM technology can enhance the effectiveness and reliability of discard data acquisition for fisheries management.

Collection of spatial information on fisheries catch and effort is essential to understanding the spatial processes of exploited population dynamics and to manage heterogeneously distributed resources and uses (Léopold et al., 2014). For this work, more than five years of EM data analyzed for hotspots and coldspots revealed that CPUE's for primary target species were not distributed similarly across the study area (Fig. 5, 8, 9, 10, 11, 13, and 15). Shallow water species, such as red grouper, showed large expanses of highly productive fishing areas (15,851 km^2), whereas deepwater species, like blueline tilefish, showed smaller areas (2,264 km^2), which lends this species to localized overfishing. Negative impacts of the recent reduction in commercial red grouper quota (NMFS[18]) and drastic cuts in gag grouper quota that industry expects to be enacted in 2023 could drive efforts offshore to these areas. Additional effort in these offshore areas could create increased interactions with Highly Migratory Species (HMS) such as scalloped hammerheads and silky sharks that show some of the highest mortality of the shark species caught on BLL gear (Table 4).

Sharks caught in this fishery are unintentional catches, with many incidents going unreported (Drymon et al.[19]), a recognized problem in many fisheries (Molina and Cooke, 2012). However, during this study, the fishermen frequently reported that shark interactions, particularly with sandbar sharks, were significantly increasing, which aligns with commercial fishermen reports included in SEDAR74-DW-32 (Drymon et al.[19]). As noted by Brewster-Geisz and Miller (2000), sandbar sharks are the most commonly observed shark species in the Gulf, despite a drastic decline from overfishing (Romine, 2008). They are prohibited from commercial or recreational harvest except through a federal research fisheries commercial harvest program (Mathers et al., 2018; SEDAR[20]). Although EM data did not show an increase in overall shark catches, a steady annual increase in sandbar shark abundance was observed (Fig. 15), validating the assertions made by participating fishermen and various sectors of the reef fish fishery (Drymon[19]; Brewster-Geisz[21]).

Fishermen's reports of an increase in sandbar shark interactions included identifying them as a primary predator causing significant depredation of catch. Depredated catch, as described in Mitchell et al. (2018) by Gilman et al. (2008), and MacNeil et al. (2009) involves catch being partially or completely consumed by an animal before it can be retrieved to the vessel. Concerns regarding the impacts of depredation, particularly from sandbar sharks, have been documented in various forums, including NMFS HMS Advisory Panel Meetings (Brewster-Geisz[21]), GMFMC meeting by NMFS HMS staff[22], public GMFMC open testimony[23], and in a report by Drymon et al.[19]. In our review of over 80,000 species-specific records of EM catch events, we found that more than 900 reef fish exhibited clear physical evidence of depredation due most likely to sharks or marine mammals. However, the exact number of depredated fish completely removed from hooks is unknown, making observed depredation rates minimum estimates. An analysis of sandbar hotspots CPUE (Fig. 16) and those of our recorded depredated species (Fig. 17) did not align to validate if this particular species of shark was driving depredation. Unfortunately, as reported by NMFS[24] the nature, extent, frequency, and geographic locations of shark and dolphin interactions in the Gulf commercial reef fish fish-

[17]SEDAR. 2017. SEDAR 50 – Atlantic blueline tilefish assessment report. SEDAR, North Charleston, SC, 542 p. (http://sedarweb.org/sedar-50).

[18]National Marine Fisheries Service (NMFS). 2022. Fisheries of the Caribbean, Gulf of Mexico, and South Atlantic; reef fish fishery of the Gulf of Mexico; Amendment 5. 87(84) (https://www.govinfo.gov/content/pkg/FR-2022-05-02/pdf/2022-09300.pdf).

[19]Drymon, M., A. Osowski, A. Jefferson, A. Anderson, D. McAree, S. Scyphers, E. Prasky, S. Swinea, S. Gibbs, M. Karnauskas, and C. Gervasi. 2022. Co-Producing a shared characterization of depredation in the Gulf of Mexico reef fish fishery: 2022 Workshop summary report. SEDAR74-DW-32. SEDAR, North Charleston, SC. 25 p., (https://sedarweb.org/documents/sedar-74-dw-32-co-producing-a-shared-characterization-of-depredation-in-the-gulf-of-mexico-reef-fish-fishery-2022-workshop-summary-report/).

[20]Southeast Data, Assessment, and Review (SEDAR). 2010. SEDAR21, Stock assessment report: HMS sandbar shark. SEDAR, North Charleston, SC, 459 p. (https://sedarweb.org/documents/sedar-21-final-stock-assessment-report-hms-sandbar-shark/).

[21]Brewster-Geisz, K. 2019. NOAA Atlantic HMS Management Division. Shark depredation, Presentation, Tab M, No. 4 (https://gulfcouncil.org/wp-content/uploads/M-4-Depredation_slides_Councils_wide.20200116.pdf).

[22]National Marine Fisheries Service (NMFS). 2022. HMS Advisory Panel Meeting. Days 1-3 Transcripts. 18-20 May 2022 (https://www.fisheries.noaa.gov/event/may-2022-hms-advisory-panel-meeting).

[23]Gulf of Mexico Fishery Management Council (GMFMC). 2022. Public testimony, 23 June 2022, 15 and 24 August 2022, and 5 October 2022 (https://docs.google.com/spreadsheets/u/3/d/e/2PACX-1vQVPwRXQn06iM-6fx44XhuB9YytIx8dsrUzuOBIhMhDgtye_1VwaMY9wtO6B4OgsZ4Bgypxmhnbjhh7x/pubhtml?gid=265588419&single=true).

[24]National Marine Fisheries Service (NMFS). 2022. Interactions between bottlenose dolphins and sharks and commercial, for-hire, and private recreational fisheries in the Gulf of Mexico and South Atlantic. Report to Congress. 29 August 2022, 49 p. (https://media.fisheries.noaa.gov/2022-08/NMFS-Assessment-Fishing-Interference-RTC-08_29_22.pdf).

ery are not fully understood. Therefore, improved data collection coverage are warranted to quantify depredation events over time and track their monetary impact. Additionally, DNA testing of depredated catch could be an asset in identifying predators, whether shark, marine mammal, or others, which can then be correlated with damage traits, including those viewed on species documented through EM. With further refined genetic techniques, it becomes feasible to identify the specific species responsible for depredation, thereby providing valuable insights to effectively inform mitigation strategies.

Additionally, there are significant management concerns regarding the observed high at-vessel mortality for hammerhead, *Sphyrna* spp., and dusky sharks, attributed to handling stress (Gulak et al., 2015; Sulikowski et al., 2020). Even though a small number of hammerhead (n = 54) and dusky shark (n = 7) interactions were documented in our study, these resulted in at-vessel mortality rates of 39% and 43%, respectively. Therefore, it is crucial to regularly obtain data on key variables associated with these capture events, including gear configuration, bait type, environmental parameters, and biology of the species of interest (Carruthers et al., 2009). Data derived from EM documenting vessel interactions with all species of hammerhead sharks were provided to SEDAR 77 by the CFEMM (Lee et al.[25]) in 2021 for consideration in the cooperative stock assessment process. Although the scalloped hammerheads are not considered endangered in the U.S. Atlantic, distinct population segments exist that NMFS considers to be endangered or threatened in other Atlantic areas (NOAA[26]). Regarding dusky sharks, the most recent stock assessment suggests this species has been so depleted in the U.S. Atlantic (including the Gulf), that the recommended rebuilding schedule may take 100 years or more to reach sustainability (NOAA, 2016; Candless et al.[27]). Due to the low numbers of these shark species encountered in the eastern Gulf BLL fishery, it is critical to maximize the documentation of these encounters to inform current and future stock assessments.

Addressing the management challenges posed by sharks, integrating EM as an additional tool to complement at-sea observer coverage to document incidental shark catches and their subsequent fate is a valuable contribution for both reef fish and HMS management. While this study has made efforts to accurately identify shark catches, with additional expertise provided by outside shark experts, it is important to note that sharks in the genus *Carcharhinus*, such as silky and dusky sharks, are difficult to discern (Beerkircher et al., 2002). Consequently, unconfirmed interactions of these species, labeled by EM reviewers under an unidentified shark grouping likely exist in the dataset. However, an advantage of EM review is that the captured video of these species events serves as permanent documentation, allowing questionable annotations to be referred to additional experts for species verification or re-examined if artificial intelligence identification tools become available in the future.

Accounting for non-target species such as sea turtles, aquatic marine mammals, and seabirds caught as bycatch during commercial fishing is a priority in fisheries management (Burgess et al., 2018; Michelin et al., 2018; van Helmond et al., 2020), with appropriate camera placement, EM has been proven to provide accurate coverage for these encounters (Michelin et al., 2018). In our studies, five sea turtles were documented, with two brought in dead and three released. No bottlenose dolphins, *Tursiops truncatus*, were observed as caught or entangled in gear, although their presence was recorded when they were observed in close proximity to a vessel. Seabirds were frequently observed actively targeting bait and discarded floating scraps from processed fish. Ten individual seabirds recorded as caught were incidental captures, often hooked during the set, resulting in a high mortality rate.

The presented work showcases the effectiveness of EM in collaboration with voluntary participants from the eastern Gulf reef fish fishery resulting in the collection of a significant amount of data on vessel operations, targeted catch, bycatch, discarded species, and interactions with protected species. Despite its limitations, EM emerges as a robust monitoring tool with solid strengths, demonstrating its potential as a valuable asset in fisheries monitoring efforts. Furthermore, this study emphasizes advantages that EM can provide if integrated into fishery dependent monitoring programs, especially in regions like the Gulf where coverage is limited. By integrating EM alongside traditional methods and integrating additional data tools as necessary within the EM platform, we can effectively address critical data gaps and bolster the overall efficacy of monitoring initiatives. However, transitioning EM from a demonstration project to an integral component of comprehensive management programs requires a distinct approach. Collaboration with management is essential to expand monitoring efforts beyond voluntary participation and ensure the acquisition of representative data. This collaborative approach is crucial for broadening the scope and depth of monitoring coverage, ultimately leading to more informed decision-making in fisheries management. In conclusion, while this work effectively showcases the capabilities of EM and

[25]Lee, M., G. Patrick, C. Neidig, and R. Schloesser. 2021. Hammerhead shark (*Sphyrna* spp.) electronic monitoring data review from the Gulf of Mexico bottom longline reef fish fishery. SEDAR77-DW05. SEDAR, North Charleston, SC, 9 p. (https://sedarweb.org/documents/sedar-77-dw05-hammerhead-shark-sphyrna-spp-electronic-monitoring-data-review-from-the-gulf-of-mexico-bottom-longline-reef-fish-fishery/).

[26]National Marine Fisheries Service (NMFS). 2020. Scalloped hammerhead shark (*Sphyrna lewini*) 5-Year Review: Summary and Evaluation. U.S. Dep. Commer., NOAA, NMFS, Off. Prot. Resour,, Silver Springs, MD, 43 p. (https://media.fisheries.noaa.gov/dam-migration/scalloped_hammerhead_5-year_review.pdf).

[27]McCandless, C. T., P. Conn, P. Cooper, E. Cortés, S. W. Laporte, and M. Nammack. 2014. Status review report: northwest Atlantic dusky shark (*Carcharhinus obscurus*). Report to National Marine Fisheries Service, Office of Protected Resources. October 2014. 72 p. (https://repository.library.noaa.gov/view/noaa/17711).

the success of industry collaboration, the next crucial step involves partnering with management to fully integrate EM into broader management frameworks, thereby maximizing its potential to enhance sustainable fisheries management practices in the Gulf reef fish fishery.

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Electronic Monitoring Performance in a Developing Deep-Set Fishery for Swordfish, *Xiphius gladius*, off California

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Introduction

Fishery monitoring and observation provide the basis for which accurate catch statistics can be generated, allowing managers to document and respond to fishery performance trends over time (Gilman et al., 2017; Bradley et al., 2019). Monitoring can provide critical fishery information such as estimates of effort, total catch and discards, protected species interactions, and issues related to fleet operations (Brooke, 2015; Wang and DiCosimo, 2019; Brown et al., 2021). Traditional forms of at-sea monitoring typically entail active fishery observation, which can be in the form of either human observers or electronic monitoring (EM) (Brooke, 2015; Moncrief-Cox et al., 2020). EM includes the use of different activity sensors and cameras positioned on vessels to remotely record fishing activity and catches (Gilman and Zimring, 2018; Carnes et al., 2019; van Helmond et al., 2019).

Given concerns over biases associated with self-reporting, human observers have long been considered the industry standard for ensuring the accuracy of fishery data collected at sea (Brooke, 2015; Curtis and Carretta, 2020). However, the high costs and logistical difficulties associated with staffing and supporting physical observers continues to offer challenges for fishing operations around the world (Haigh et al., 2002; Mangel et al., 2013; Alfaro-Córdova et al., 2017). Additionally, issues related to personal safety and liability as well as intimidation and bribery may collectively contribute to the difficulty associated with staffing fishery observers and collecting accurate information from open ocean fisheries (Brown et al., 2021; Belhabib and Le Billon, 2022).

To help increase coverage rates and mitigate some of the issues that are often associated with human observation, several forms of EM technology have been developed for meeting regulatory and compliance needs (Gilman and Zimring, 2018; reviewed by van Helmond et al., 2019). EM has now been trialed and implemented on over 1,000 vessels worldwide, with the majority of the innovation to date focused on larger vessels from industrialized fleets (Carnes et al., 2019; Itano et al., 2019; Brown et al., 2021). More recently, effort has been directed towards meeting the monitoring needs of smaller, artisanal operations which often go overlooked due to their low relative value compared to more industrialized operations (Bartholomew et al., 2018; van Helmond et al., 2019). This is critical considering that over half of the world's fish production comes from small-scale artisanal fleets, many of which continue to lack any form of monitoring (Bartholomew et al., 2018; Arthur et al., 2022).

Identifying uniform monitoring solutions in many small-scale fisheries can be difficult due to variability in fishery revenue potential, vessel sizes, and differences in observational or monitoring objectives. Fortunately, EM technology has progressed sufficiently such that monitoring platforms can be designed to address a wide range of fishery objectives as well as regulatory and compliance needs (reviewed by van Helmond et al., 2019).

Despite these advancements to date, building an effective EM platform still requires custom system tailoring to meet vessel specifications as well as the monitoring objectives of the fish-

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ABSTRACT—Electronic monitoring (EM) systems have been developed to facilitate fisheries data collection in many industrialized fleets but remain less common onboard smaller vessels. This work tested the performance of small-vessel EM systems to meet the reporting objectives in the California deep-set buoy gear (DSBG) fishery for swordfish, Xiphias gladius. *Two EM service providers were contracted to develop monitoring platforms, which included both 1- and 2-camera systems. Initial testing was performed onboard a research vessel to identify functional configurations and refine data collection procedures prior to commercial testing.*

Commercial trials were performed onboard four vessels during the 2020–21 fishing seasons, resulting in 126 electronically monitored set days. Fishery performance metrics from EM records were compared between service providers, fishing logbooks, and records from NOAA-certified observers. Catch comparisons yielded full agreement (100%) for both target (109/109) and incidental catch (9/9) between the 2-camera EM system and logbook records. The 1-camera system yielded 88% agreement (52/59) for target catch and 14% (1/7) for incidental catch. Unquantified catch events occurred outside the 1-camera field of view onboard larger vessels with independent hauling and catch-processing areas. Collectively, this study demonstrates that EM can be used to accurately document fishery catch statistics in the developing commercial DSBG fishery.

ery (i.e., catch limits, bycatch quantification, logbook verification; Bradley et al., 2019; Moncrief-Cox et al., 2020). Field testing and verification is also needed to demonstrate EM effectiveness and relative performance compared to human observers (Gilman et al., 2020). This is especially important if the fishery is new and its characteristics have not been previously documented.

California Deep-set Fishery

Research and testing under Exempted Fishing Permit (EFP) status has led to the recommendation[1] and recent authorization[2] of deep-set buoy gear (DSBG) and linked buoy gear (LBG) under the West Coast Fishery Management Plan for Highly Migratory Species (Sepulveda and Aalbers, 2018; Sepulveda et al., 2024). During this project period, the DSBG fishery was operating under exempted status through the Pacific Fisheries Management Council (PFMC).

The DSBG fishery is artisanal in nature, with vessels typically deploying between 10 and 30 hooks daily, targeting swordfish below the thermocline during the day. The fleet is composed of vessels that range in size from 8–20 m, with trip durations ranging from 1 to 10 days. Fishery performance to date suggests DSBG to be highly selective for swordfish (>90% target catch), with non-target catch commonly released alive (Sepulveda and Aalbers, 2018; Aalbers et al., 2021; Sepulveda et al., 2024).

During this work, fishery monitoring requirements were set by regional managers (i.e., PFMC, NOAA) and met with physical observers staffed and provided by the NMFS West Coast Observer Program[3]. Although the DSBG fishery has been characterized as being low-impact and artisanal in nature, swordfish operations (i.e. pelagic longline, drift gillnet) in the north Pacific have had a history of protected species interactions, a factor that continues to support the use of physical observers and at-sea monitoring efforts (Carretta et al., 2004; Urbisci et al., 2017).

Given the lack of published data on the use of EM in any buoy-based fishery, we performed a pilot study to assess EM performance compared to both human observation and fishing logbook data (Murua et al., 2020; Brown et al., 2021). The objective of this study was to evaluate the suitability and efficacy of small-vessel EM systems as well as identify the system requirements necessary to effectively monitor the developing DSBG fishery.

The DSBG fishery is well suited for EM given that 1) smaller deep-set vessels have reduced observer accommodations, 2) daily fishery revenues are often insufficient for supporting human observer costs, and 3) fishery participants have voiced interest in exploring non-human monitoring options to reduce the financial and space-sharing burden associated with carrying human observers. Because physical observation was mandated during exempted fishing trials and gear development efforts (Bonito et al., 2022), this work capitalized on the data collected by human observers and summarized by the NMFS West Coast Region. Considering that neither of the small-vessel EM platforms utilized in this study had previously undergone extensive field tesing, this work documented fishery performance metrics (i.e., fishing locations and times, catch and effort) from both 1-camera and 2-camera data collection systems mounted onboard DSBG vessels targeting swordfish off southern California.

Methods

Location and Study Design

All research and EFP sets were performed within a portion of the Southern California Bight extending from approximately Point Conception to the U.S.–Mexico border. Because the EM technology tested in this study had not been used in any commercial application off California, testing was carried out in two phases. Phase I was performed aboard a research vessel to develop and refine the EM systems, followed by Phase II testing aboard active DSBG vessels to evaluate system performance during commercial fishing activities.

Gear configurations used during the study included both DSBG and LBG, which use the same basic hauling platform and have been shown to result in similar catch composition (Sepulveda and Aalbers, 2018; Aalbers et al., 2021; Sepulveda et al., 2014, 2024). Phase I research sets were performed under a Scientific Collection Permit (SCP) issued through the California Department of Fish and Wildlife (Specific Use Permit ID S-183330009-19106-001) as well as a Federal Letter of Acknowledgment issued through NOAA's NMFS West Coast Region.

EM Service Provider Selection

Independent surveys with EM service providers were conducted to identify a suitable small-vessel system for the study. The principal factors considered were cost and suitability onboard small vessels, as the fishery is artisanal in nature and the vessels are relatively limited in size (8–20 m). Two EM service providers were chosen to participate in the study, Saltwater Inc. (SI) and Shellcatch Ltd. (SC)[4], both of which had a history of performance in the EM field and were willing to work with the team to tailor a platform for the California deep-set fishery. Both providers offered comparably priced small-vessel systems and incorporated

[1]National Oceanic and Atmospheric Administration. 2019. Federal Register 50 CFR Part 300, 84(42):7,323–7,325. (Avail. at https://www.federalregister.gov/documents/2019/03/04/2019-03493/fisheries-off-west-coast-states-highly-migratory-fisheries-amendment-6-to-fishery-management-plan). Accessed 11 July 2023.

[2]National Oceanic and Atmospheric Administration. 2023. Federal Register 50 CFR Part 660, 88, (88): 29544-29545. (Avail. at https://www.federalregister.gov/documents/2023/05/08/2023-09748/fisheries-off-west-coast-states-highly-migratory-fisheries-amendment-6-to-the-fishery-management). Accessed 11 July 2023.

[3]National Marine Fisheries Service West Coast Region Observer Program (avail. at https://www.fisheries.noaa.gov/west-coast/fisheries-observers/west-coast-region-observer-program). Accessed 11 July 2023.

[4]Mention of trade names or commercial companies is for identification purposes only and does not imply endorsement by the National Marine Fisheries Service, NOAA

data review services using their own respective software packages.

Saltwater Inc. (SI)

Although previous EM systems have been developed by SI for use in other commercial fisheries (i.e., pot, trawl, longline; Buckelew et al., 2015), the small-vessel system used in this study had not been tested in any other commercial application. During Phase I trials, SI initially provided a 1-camera system that was replaced in June, 2021 with a dual-camera system using hardware adapted from a commercially-available vehicle dash camera[5]. The resulting SI small-vessel EM platform used a custom waterproof digital video recorder (DVR) that interfaced with a GPS receiver and two independent camera heads extending from the unit with cable lengths of 1.5 and 3.0 m.

For purposes of comparison, one camera was mounted directly adjacent to the SC platform, while the other was positioned to maximize coverage based on the unique vessel layout. The DVR was mounted in an accessible area and wired with constant 12-V power from the vessel battery bank. An additional lead was wired to the main engine ignition switch such that recording was triggered only when the ignition was engaged, in order to conserve memory and minimize review time. Video was recorded at a rate of 30 frames s^{-1} and a resolution of 1080 pixels from both cameras with a 120°x100° field of view (FOV). Data retrieval required the physical exchange of 256-GB micro SD cards. Data were subsequently processed using the SI import tool and analyzed using the SI open-source review software. Datasets were stored on local hard drives, which were sent via courier to the service provider for external review.

Shellcatch Ltd. (SC)

The SC system used a single, self-contained, watertight camera unit that was coupled with the SC Virtual Observer app. The system was designed for monitoring artisanal vessels and has been previously used in small-scale fisheries in Latin America (Bartholomew et al., 2018). The SC system featured a built-in GPS receiver and captured still images at a rate of 20 frames min^{-1} with a 62°x48° FOV and a resolution of 1280 x 720 pixels.

Recording was automatically initiated when the integrated geographic restriction feature (geo-fence) detected vessel movement beyond a programmed radius surrounding its home port. Camera positioning and recording status could be verified in real time through the Virtual Observer app accessed from a personal electronic device (i.e., phone or laptop). Image and GPS data were stored internally until a bulk wireless data transfer was performed.

To facilitate data transfer, SC camera units interfaced with a Virtual Observer uploader tool, which required a wired internet connection. Data upload procedures required vessel operators or crewmembers to remove camera units from their mounted position and link the system with the uploader unit. When the uploader detected a camera unit nearby, wireless data transfer was automatically initiated at a rate of 4 MB/s from the camera unit to the uploader, which then transferred the data to the SC cloud-based server. Camera and uploader status were monitored online through the SC dashboard portal to confirm data transfer prior to remounting on the vessel.

Research Trials (Phase I)

Phase I testing was conducted onboard the R/V *Malolo* (14 m) during the 2020–21 fishing season to evaluate camera functionality, recording resolution/frame rates (i.e., still frames vs. continuous video), on/off protocols (ignition triggered vs. geo-fence), as well as data acquisition and transfer procedures. Both EM platforms were mounted, side by side, to the aft rail of the RV *Malolo* control tower at a distance of 12 m above the back deck to provide a view of the entire working area. The Pfleger Institute of Environmental Research (PIER) vessel had a similar layout as the participating commercial vessels and used the same hauling platform when fishing with either DSBG or LBG. All catch was tagged and released during research trials, thus direct comparisons of catch data were not conducted between experimental phases because of differences in fish tagging and handling procedures. Research sets were reviewed by the PIER team using corresponding image review software. Research data and review materials collected during Phase I were subsequently used to train EM service provider analysts and to create a set of standardized review protocols. Findings from Phase 1 were also used to determine preferred camera configurations and recording protocols for subsequent commercial trials. Phase 1 trials were initiated with an SI 1-camera system that was replaced in June, 2021 with an SI 2-camera system, which was selected for use during Phase 2.

EFP Trials (Phase II)

Upon completion of the research trials, Phase II testing was conducted during the 2021 fishing season onboard four commercial fishing vessels participating in deep-set EFP trials. Prior to installation, the team met with all cooperative fishermen to coordinate setup, discuss confidentiality clauses, and train crew on camera functionality, field diagnostics, application features, and data handling protocols. Cooperative fisherman participation was largely supported by their collective interest in developing ways to reduce the financial and space-sharing burden associated with accommodating physical observers. Both systems were mounted and configured as instructed by service providers and positioned side by side within the tower of each participating vessel (Fig. 1). Because EFP vessels were approved to use either DSBG or LBG configurations, mounting positions were selected to encompass both gear deployment and haul-back stations in addition to where catch was processed. However, the distance between hauling and processing sta-

[5]Innovv K2 MotoCam. (Avail. at: https://www.innovv.com/innovv-k2). Accessed 11 July 2023.

Figure 1.—Commercial swordfish vessel outfitted with two electronic monitoring (EM) platforms, including both a 1-camera system from Shellcatch Ltd (solid red circle) and a 2-camera system from Saltwater Inc (dashed red circle) during experimental deep-set fishing trials within the 2020–21 fishing seasons.

tions on the two larger vessels precluded capture of the entire working deck from a single camera angle.

Participating vessels ranged in size from 11.6 to 19.5 m, (mean=15.4 m), operated by a captain and 1–3 crewmembers. Vessel selection criteria were consistent with established EFP terms and conditions, which specified that vessel captains must: 1) maintain accurate logbook entries for every set, regardless of observer presence; 2) carry a NOAA-certified observer on any trip that is requested of them; 3) notify PIER prior to departure on any trip and maintain consistent communication through daily check-in reports; and 4) immediately report the catch or interaction with any protected species. Vessel operators were compensated for the added undertaking of carrying both physical observers and housing EM systems as well as coordinating with team members for installations and data transfer. During Phase II trials, all swordfish caught were retained by the vessel for commercial sale and all incidental catch was released.

Observer and Logbook Records

Fishery observer training and data management was facilitated through the NMFS West Coast Region (WCR) Observer Program, while observer employment and placement onboard EFP vessels was coordinated through the contracted agency Frank Orth and Associates (FOA; Long Beach, CA). EFP vessel operators were required to contact the FOA observer coordinator 48 h prior to trip departure for determination of whether a physical observer would be placed onboard the pending trip. Because of altered staffing protocols due to COVID-19 restrictions, physical observers were not consistently available during the EFP trials.

Following trip debriefing, physical observer data were delivered to the WCR office for entry into an HMS database. Observer data records for all 2021 trips aboard EFP vessels that carried EM systems were requested by PIER for subsequent comparisons

with output files from EM review and fishing logbook data. As described by Sepulveda et al. (2024), fishing logbook records were compared with daily check-in reports, state landing receipts, and observer data when available. Any discrepancies were identified for further evaluation.

Review Protocols

Prior to initiation of Phase II commercial trials, review training sessions were conducted with both service providers using the Phase I research trial data. Reviewers were given an overview of the gear types, configurations, and common species encountered in the fishery. Specific signatures, such as rapid changes in vessel speed or direction, were identified to help expedite the review process and better pinpoint significant events. The research team then worked directly with image analysts to develop consistent review protocols designed to capture the same information recorded by physical observers.

Recorded information included: 1) departure and arrival dates and locations; 2) number of days fished; 3) gear types deployed; 4) quantity of gear set; 5) catch composition; 6) catch condition and disposition; and 7) system performance issues. Event annotations were formatted with the intent to generate output files that were comparable with physical observer data, allowing for compatibility with existing NMFS databases. Because each system used a different review platform, the features of each are outlined individually below.

SI Review Platform Features

The SI system used the open-source O2 Review software package. The software utilized video data that was stored locally, either on a PC or external hard drive. Raw camera data underwent an upload process using the SI import tool, which formatted video files to be recognized by the review software. The user interface displayed video playback, a digital chart containing GPS coordinates and track lines, and a timeline displaying vessel speed. Information was displayed in separate windows to allow analysts to review multiple angles of video footage along with concurrent vessel speed and movements (Fig. 2A). The video playback rate was adjustable between 0.25 and 128x actual speed, allowing reviewers versatility in scrutinizing capture events and bypassing periods of inactivity. Annotations were made on the O2 review timeline and included fishing information as either point (i.e., observed swordfish capture event) or duration (i.e., linked buoy gear set) events, and trip information such as departure and arrival date and location. Any annotations made during the review process were saved in an associated data folder that could be viewed or modified on other devices.

SC Review Platform Features

The SC system used an online review portal (Shellcatch Review Dashboard), that accessed cloud-based image data that was stored on an SC server. The review interface displayed images, vessel speed, and GPS data on a single webpage that could be viewed at rates from 0.5 to 4x actual speed (Fig. 2B). Specific events (i.e., gear deployment, target species catch, bycatch) were annotated within designated fields adjacent to the timeline, such that exported data was properly categorized and associated with a specific time and location. All annotations were saved to the online dashboard for subsequent access by other reviewers.

Data Analysis

PIER analysts independently reviewed all Phase I research sets in addition to validating both SI and SC review data against fishing logbooks and observer records during Phase II trials. Daily catch and effort statistics from logbook and observer reports were compared to image review output data for all vessels and dates with EM records. Any differences between daily catch records were individually assessed to determine the probable cause of discrepancies.

Catch discrepancies were attributed to either a reviewer, camera, or logbook entry error. Loss of data caused by human oversight resulted primarily from poorly repositioning cameras after servicing, improper wiring, failure to download cameras on schedule, and damaging memory cards. Data loss attributed to human oversight contributed to discrepancies in the number of entire sets recorded by each monitoring system. The level of physical observation was also compromised during two trips that human observers became sea sick or needed to return to port, as well as on trips when observers were not available or logistical constraints prevented placement.

EM performance metrics used to evaluate specific features of each system included: 1) number of cameras, 2) type of imagery (still vs. video), 3) mode of data storage and transfer (cloud-based vs. data storage cards), 4) initiation triggers, and 5) review software package. Overall storage capacity was determined as the number of continuous recording days available for both EM systems. The level of fishing effort was estimated from the number of individual buoys or linked sections deployed along with the soak duration for each DSBG or LBG set. Calculations of hook-soak hours were based on the number of hooks deployed per buoy line or linked section and the daily fishing duration. The time required to review each set was documented for a subset of trips to estimate average daily review times.

Results

Phase I

All experimental sets aboard the PIER research vessel were made between Santa Cruz Island and the U.S.–Mexico border and out 60 km from the southern California coastline. During Phase I, a total of 23 research sets were conducted between 3 September 2020 and 8 June 2021 with EM systems onboard. The SC 1-camera system monitored all 23 sets, while the initial SI 1-camera system monitored 5 sets during Phase 1 prior to transitioning to the SI 2-camera platform. Image review outputs matched actual catch values for 100% of the target catch events on

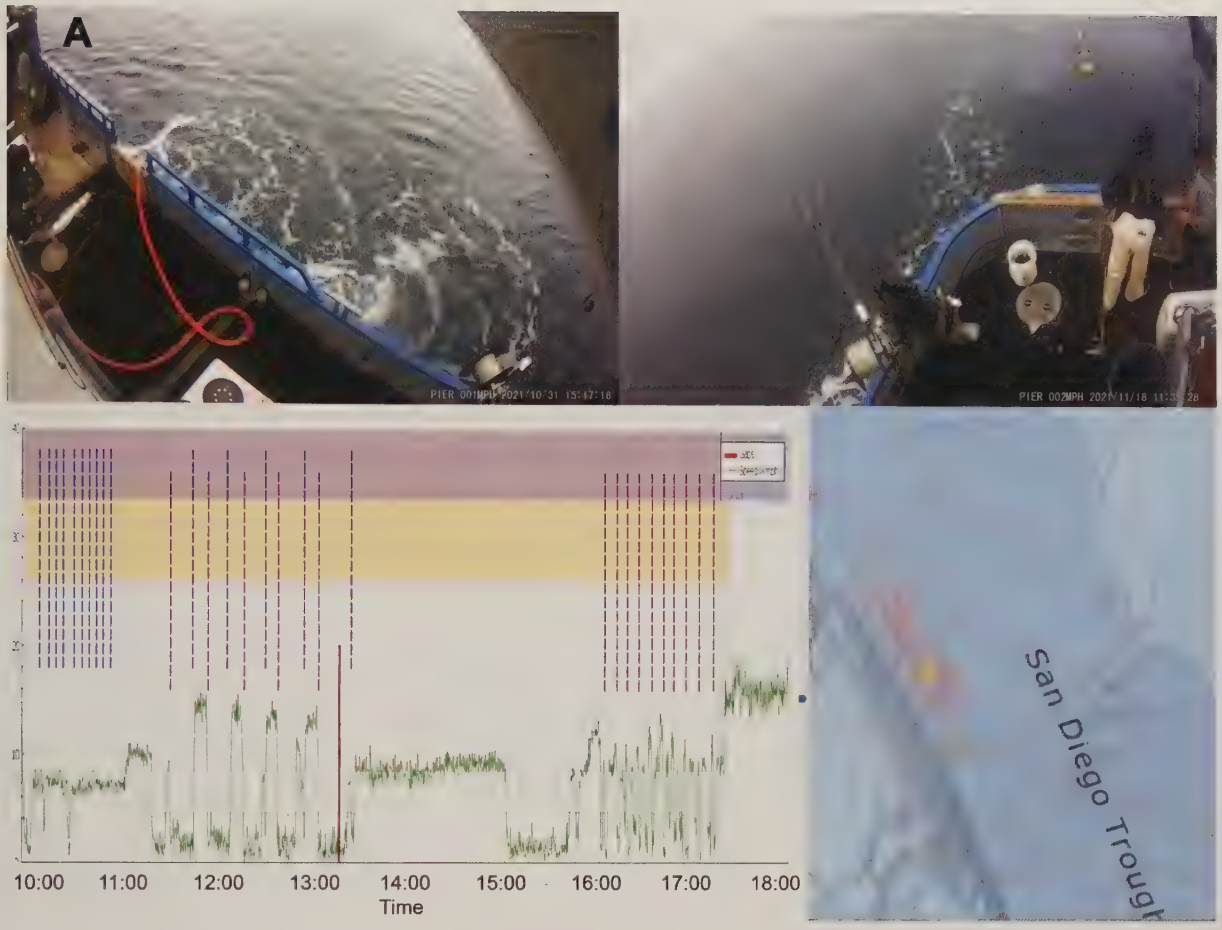

Figure 2.—Screenshot of the open-source O2 Review software package by Saltwater Inc. (A) and the electronic monitoring review software developed through the Shellcatch Review Dashboard (B), both featuring expandable windows for viewing overhead camera view of the working deck, as well as the image timeline, vessel trackline, and speed.

both the SC 1-camera (20/20) and the initial SI 1-camera (6/6) systems.

During the same sets, there were 10 total non-target interactions consisting of three different species, including the shark bigeye thresher, *Alopias superciliosus*; pelagic stingray, *Pteroplatytrygon violacea*; and blue shark, *Prionace glauca*. The SC system was operational during all 10 of the non-target capture events and EM records matched actual catch values for 70% (7/10) of the events. The SI system was operational during 9 of the non-target interactions and EM records matched actual catch values for 89% (8/9) of the events. Inability to identify catch was attributed to either subsurface release or catch being released outside of camera FOV.

Phase II

From 10 July through 3 December 2021, four participating vessels conducted 33 deep-set fishing trips, ranging in duration from 1 to 8 days (mean=4.2 d). A total of 126 sets were successfully recorded by either one or both of the EM systems, while physical observers documented 23% of the sets (29/126; Table 1). EM and observer records yielded good agreement (99% agreement, 139/140 events) with logbook data for both target and incidental catch. Observer records, logbook data, and image analysis confirmed the catch of just three species, swordfish (Fig. 3 A–C), bigeye thresher (Fig. 3D), and oilfish, *Ruvettus pretiosus*. The results of Phase II trials are presented individually below based on the presence of either 1 (SC) or 2 (SI) cameras.

The SC 1-camera platform successfully recorded 78 set days during Phase II trials. EM data showed agreement with logbook records for 88% (52/59) of the total swordfish catch. Of the 29 days in which a physical observer was

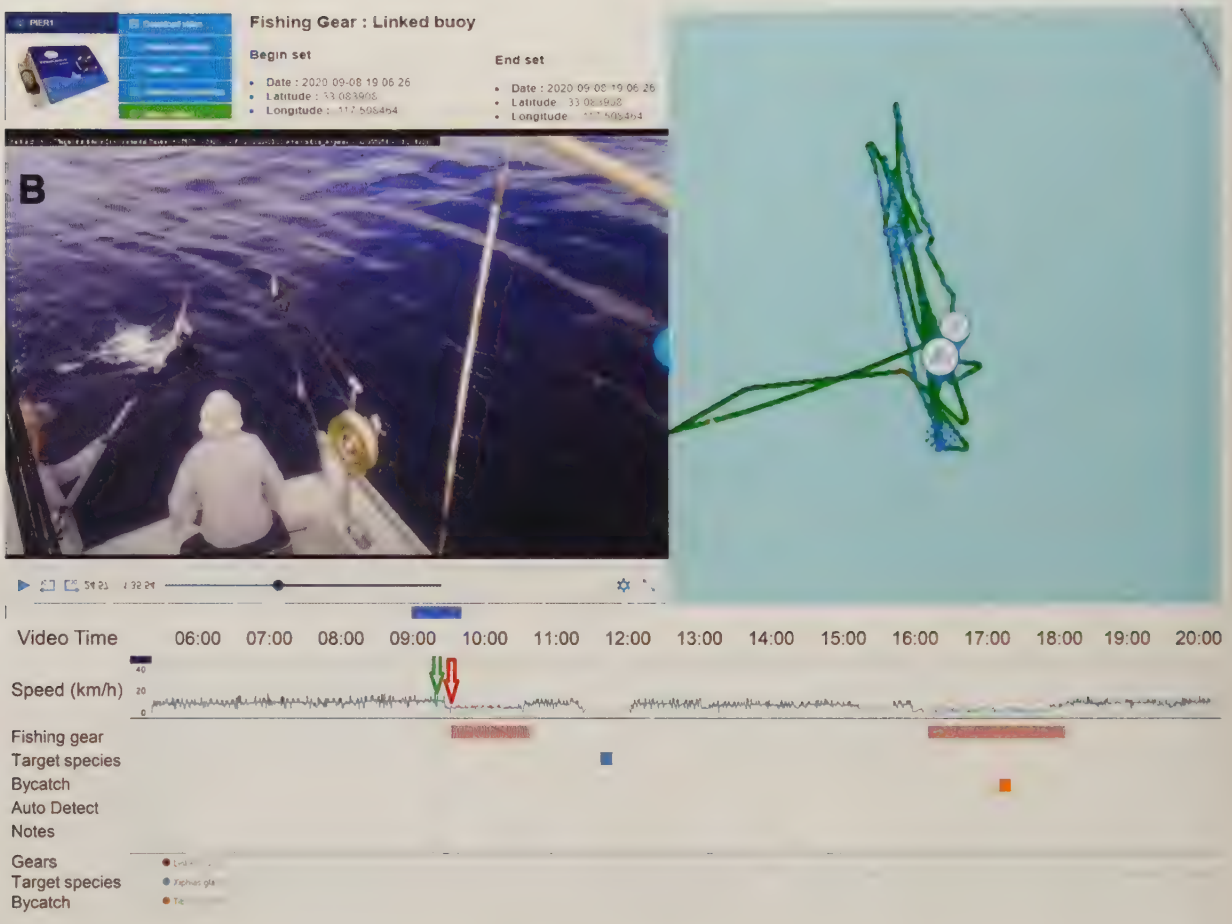

Figure 2.—*Continued.*

also present, observer records verified that the SC 1-camera system identified 21/28 target catch events (75%). All 7 of the swordfish not documented by EM were caught aboard larger vessels with processing areas outside of the camera FOV.

For the incidental catch, EM data yielded agreement with logbook records for 14% (1/7) of the total non-swordfish catch. Five of these events were verified by the presence of a physical observer and two were based on fishing logbook records and verified by the other camera system. The incidental catch events not documented by the SC 1-camera system occurred outside of the camera FOV.

The SI 2-camera system successfully captured 97 set days during Phase II trials. EM data showed agreement with logbook records for 100% (109/109) of the total swordfish catch. Of the 22 days in which a physical observer was also present, observer records verified 15 of the target catch events. For incidental catch, the SI 2-camera system also yielded agreement with logbook records for 100% (9/9) of the total non-target events, with three of these events verified by the presence of a physical observer. One of the bigeye thresher catch events that was documented by both the EM system and physical observer was not present in the logbook (i.e., human or logbook entry error).

Table 1.—Reported catch numbers and verification levels documented by physical observers, cooperative fishermen, and two electronic monitoring (EM) systems employed during field trials onboard four commercial deep-set fishing vessels targeting swordfish off California.

Observation mode	# non-target events recorded vs. actual	% non-target verification level	# target events recorded vs. actual	% target verification level	Overall performance level
Human observer	5/5	100.0%	26/26	100.0%	100.0%
Fishing logbook	10/11	90.9%	129/129	100.0%	99.3%
1-camera EM	1/7	14.3%	52/59	88.1%	80.3%
2-camera EM	9/9	100.0%	109/109	100.0%	100.0%

Figure 3.—Captured images from electronic monitoring equipment outfitted on each of the four commercial deep-set buoy gear EFP fishing vessels used in this study to monitor target catch rates of *Xiphias gladius* (A, B, C) and non-target interactions prior to release, including *Alopias superciliosus* (D), which is identifiable by the elongated upper lobe of the caudal fin visible above the surface of the water.

EM System Performance

The average review time for each set day was ~60 min for the SC platform and ~30 min using the SI platform. On average, the SC system was able to record ~16.9 d of continuous video footage before bulk data uploads were required, and the SI system was able to record ~15.5 d of continuous video footage before SD card data transfers were needed.

Data retrieval and download varied between service providers and required the physical manipulation of either the SD card or camera unit. Secondary review was used to verify discrepancies between data sources for both target and incidental catch for both service providers. The SC system required secondary review for ~40% of the catch events (target and non-target) while the SI system required clarification for ~10% of the events.

Discussion

This study documented the performance of EM technology in monitoring catch, bycatch, and effort in a developing west coast fishery targeting swordfish. The study used two independent EM service providers and compared findings from small boat EM systems that were based on either a 1- or 2-camera platform. Comparisons of EM records with observer and fishing logbook records show that as long as the camera FOV adequately covered both the hauling and fish processing areas, the system was able to document 100% of the catch events.

For the SC 1-camera system, larger vessels with split working areas proved to be difficult to monitor with 100% confidence. Data from the initial research testing suggested that future handling protocols (i.e., bringing catch to the surface prior to release) may be useful for identification of non-target catch, regardless of the number of cameras used. Although this study showed that EM can accurately quantify fishing activity and catch composition in the developing deep-set fishery for swordfish, we suggest additional refinements that may help improve performance for any future commercial application.

EM-System Performance

This study simultaneously tested two comparably priced EM systems that differed in terms of their number of cameras, mode of data retrieval, and

review software. Both platforms were initially tested onboard the R/V *Malolo* to assess system performance and allowed both service providers to address concerns or make adjustments prior to commercial testing. Based on findings from research trials, SI transitioned from its original 1-camera system to a 2-camera platform, which increased camera FOV and overall performance during Phase II trials.

Despite differences in system attributes, the primary factor that impacted monitoring efficacy was camera FOV. Previous studies also considered camera FOV to be a limiting factor in EM performance and similarly found that an individual camera view may be obstructed from portions of the working deck (Ames et al., 2005; Ames et al., 2007; Bartholomew et al., 2018). Because swordfish were often hauled onboard and processed in a different location from where non-target species were released, catch events may go undetected when the camera FOV did not cover both the hauling station as well as the deck door where catch was brought onboard. The SI 2-camera system used in this study had a collectively wider FOV and was more effective at capturing both the hauling and processing areas aboard larger vessels. Although the reduced FOV of the 1-camera systems resulted in a decreased number of documented catch events on the larger vessels (>17 m), the single camera was adequate for monitoring smaller platforms.

EM review accuracy (i.e., actual vs. observed interactions) and the amount of time required for review both improved from features of the SI 2-camera system (i.e., continuous video, the use of a wider FOV, and second camera angle). Analysts reported that reviewing continuous video, higher-resolution images, a wider range of playback rates, and multiple camera angles were advantageous over reviewing lower-resolution images with a reduced frame rate from a single-camera angle. However, based on the records from the smaller vessels, it was apparent that the fishing activities can be monitored successfully using still frames (20 fpm), as long as the FOV covered the entire working deck.

The different forms of data storage and transfer between systems each had benefits as well as disadvantages. While cloud-based platforms did not require physical data storage, difficulties associated with removing the SC camera between trips for data transfer resulted in poor re-positioning and missed data collection opportunities in this study. The retrieval of cloud-based data for review also involved loading and buffering, which added to total review times. The use of physical data-storage cards allowed data transfer without removing systems from their mounts; however, managing the transfer of cards between trips was also problematic and resulted in recording gaps and data loss. For example, a micro-SD card was broken during transfer resulting in the loss of data from 11 sets, a problem that could be minimized with reduced data handling.

Designing monitoring systems that incorporate backup mechanisms and minimize handling may considerably enhance data collection potential through reducing human oversight. Data stored locally on physical hard drives resulted in more efficient image playback and expedited review times. For both systems, limited storage capacity (~16-d recording time) was also problematic given that opportunities to transfer data between trips were not always convenient or possible, particularly when swordfish catch rates were high.

For example, fishing vessels rarely remained in port for more than 24 h when fishing was good and were commonly offloaded at odd hours, which impacted opportunities for data transfer between trips and led to data loss from human oversight during periods when catch rates were heightened. Although the 2 small-vessel EM systems used different start and stop mechanisms, both the geo-fence (SC) and ignition trigger (SI) were effective at reducing the overall amount of data recorded during periods of inactivity, which expedited review and extended system storage capacity.

EM Review Comparisons

Although not a principal goal of this study, a general comparison of the two service provider data review platforms was conducted to identify features that may be advantageous in a future fishery application. For both EM packages, image analysts used specific patterns found in the GPS tracks and vessel speed to identify capture events and reduce review times. Capture events were typically associated with a prolonged period of reduced vessel speed and changes in heading, whereas periods of transit or gear monitoring were more linear and at a consistent vessel speed (Fig. 4). Findings suggest that the development of machine learning algorithms may be incorporated into future review efforts, a strategy that may substantially reduce the time and costs associated with data review (Carnes et al., 2019; Qiao et al., 2020; Kay and Merrifield, 2021).

Review processes may also be streamlined by additional features used to detect fishing activity, such as optical sensors for spool or deck movement (Gilman et al., 2019). Review analysts noted that having a greater range of playback speeds (0.1–128x) and window sizes also expedited review times and often minimized the need for secondary analyses. Review times were also delayed due to limitations associated with remotely accessing data that was stored on a cloud-based server, an issue that may be overcome with access to data from local hard drives.

System Improvement Recommendations

Although this work has demonstrated the capacity of EM to successfully record and document deep-set fishing activity, we identified several areas in which a future system could be improved. A source of error in this study resulted from the frequent manipulation of both camera systems during data retrieval and transfer procedures. Given the difficulties and time associated with data transfer in addition to the loss of data caused by human oversight, an optimized commercial system

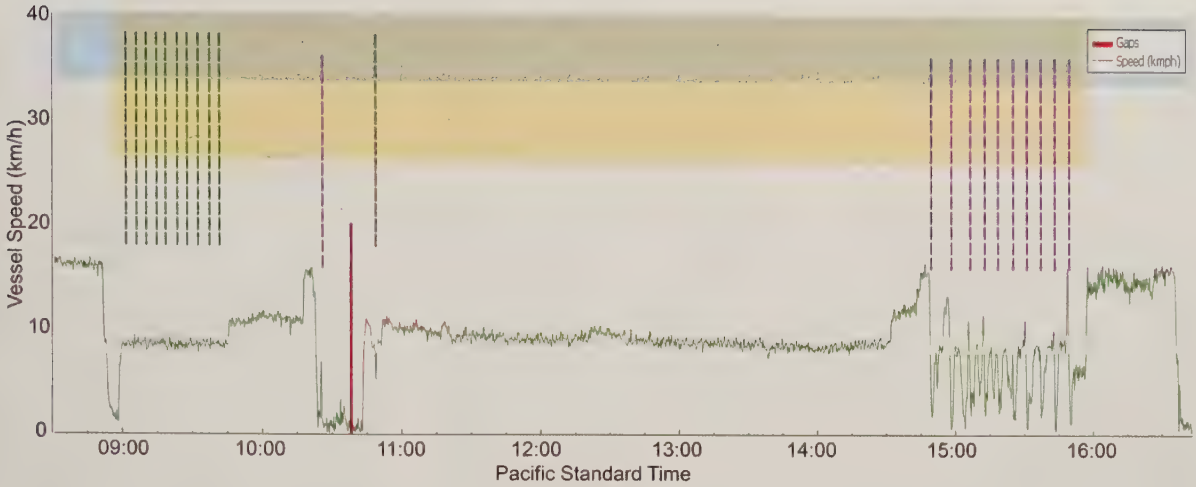

Figure 4.—Expanded timeline illustrating identifiable changes in vessel speed for an annotated catch event (red vertical line) that was used by image analysts to expedite review of EM footage using open-source O2 Review software package developed by Saltwater Inc. Green vertical lines represented gear deployment and purple lines represent gear retrieval events, while red shaded area represents duration of fishing effort.

should enable increased storage capacity to minimize camera handling and data transfer to the extent possible. Another option may be to integrate satellite-based data uploading capabilities for the remote transfer of data to a cloud-based storage network. Although satellite-based data transfer onboard small-vessels has previously been cost prohibitive, many west coast commercial vessels have recently identified more affordable at-sea connectivity options. Both initiation triggers used in this study prolonged periods between data transfers; however, future systems may benefit from a combination of features to help reduce the amount of non-essential activity recorded.

To account for FOV limitations, we would recommend the use of a 2-camera system on larger vessels, especially those with separate hauling and fish processing areas. Although efforts were made to minimize obstructions by mounting cameras in elevated positions, a 2-camera system provided a more unobstructed view of the entire working area onboard larger vessels. Increased FOV from more than one camera angle improved the proper identification of non-retained catch. However, to further enhance species identification, we recommend that all non-retained catch be held at the surface of the water prior to release. Modified handling protocols may increase EM identification accuracy by preventing the underwater cut off of branch lines (Emery et al., 2018; Carnes et al., 2019), a change that could also benefit physical observers that similarly may not be able to identify underwater or distantly released catch. The use of overhead or underwater illumination may also be useful to better identify catch released under low-light conditions.

DSBG Fishery Future Use of EM

Although the deep-set fishery has demonstrated a high level of selectivity for swordfish, the new fishery operates within the same areas that previously generated concern over protected species interactions (Carretta et al., 2004; Martin et al., 2015; Sepulveda and Aalbers, 2018). Considering the strong environmental standards among state and federal agencies, it is likely that an authorized west coast fishery will require some level of observation. Findings presented both here and in previous studies (Ames et al., 2007; Kindt-Larsen et al., 2012; Bartholomew et al., 2018) show that EM systems provide a reliable option for collecting catch information from vessels that may not be suitable for human observation due to a variety of reasons (i.e., size, safety, distance from port).

Considering the logistical challenges associated with physical observation onboard small-vessels (i.e., space limitations, insufficient bunk accommodations, and increased galley expenditures), it is evident that EM can benefit and compliment current fishery observation programs. Additionally, the use of EM can significantly reduce the overall cost of observation. Because the growing DSBG fleet is comprised of a wide range of vessel types and sizes, some of which are not suitable for accommodating physical observers overnight, the incorporation of EM as a future observation option may help provide an equitable observation platform for the developing west coast fishery or during other proposed EFP activities using different gear configurations. Lastly, given increased hesitancy to share personal space since the Covid-19 pandemic, EM may be more readily embraced and preferred by fishermen in the future (Erasmus et al., 2022).

Other Small-Scale Fisheries Future Use of EM

As EM technology improves and becomes more cost effective, the challenges associated with monitoring artisanal fishing operations are lessened. This study demonstrated the successful deployment and testing of two EM systems on small-scale commercial vessels (<12 m) and suggests that comparable designs could be used to effectively monitor fleets of even smaller boats. Considering that artisanal fisheries comprise a significant portion of the global bycatch concerns and total commercial harvest levels, the opportunity to document small-scale fishing activities may provide managers with the tools needed to better understand the vast numbers of unmonitored vessels that operate around the world.

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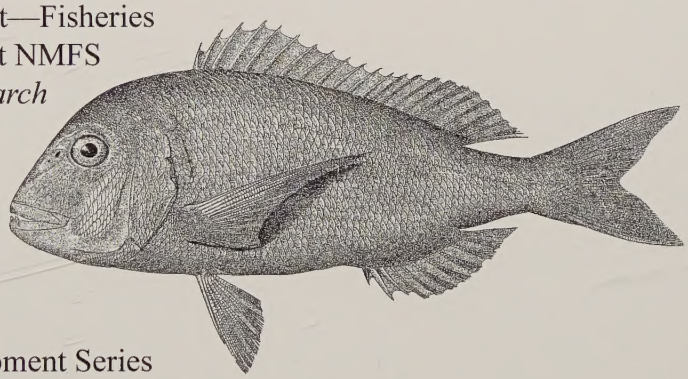

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